

# Post-Release Attributes and Survival of Hatchery and Natural Fall Chinook Salmon in the Snake River

**Annual Report  
2000 - 2001**



DOE/BP-00005362-1

February 2003

This Document should be cited as follows:

*Tiffan, Kenneth, Dennis Rondorf, William Connor, "Post-Release Attributes and Survival of Hatchery and Natural Fall Chinook Salmon in the Snake River", Project No. 1991-02900, 197 electronic pages, (BPA Report DOE/BP-00005362-1)*

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P.O. Box 3621  
Portland, Oregon 97208

This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views in this report are the author's and do not necessarily represent the views of BPA.

**Annual Report 2000-2001**

**POST-RELEASE ATTRIBUTES AND SURVIVAL OF HATCHERY  
AND NATURAL FALL CHINOOK SALMON IN THE SNAKE RIVER**

**February 2003**



POST-RELEASE ATTRIBUTES AND SURVIVAL OF HATCHERY AND  
NATURAL FALL CHINOOK SALMON IN THE SNAKE RIVER  
ANNUAL REPORT 2000-2001

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Project Number 91-029  
Contract Number DE-AI79-91BP21708

<http://www.efw.bpa.gov/Environment/EW/EWP/DOCS/REPORTS/GENERAL>

February 2003

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## **Executive Summary**

This report summarizes results of research activities conducted in 2000, 2001, and years previous to aid in the management and recovery of fall chinook salmon in the Columbia River basin. The report is divided into sections and self-standing chapters. For detailed summaries, we refer the reader to the abstracts given on the second page of each chapter. The Annual Reporting section includes information provided to fishery managers in-season and post-season, and it contains a detailed summary of life history and survival statistics on wild Snake River fall chinook salmon juveniles for the years 1992-2001. The Journal Manuscripts section includes complete copies of papers submitted or published during 2000 and 2001 that were not included in previous annual reports. Publication is a high priority for this project because it provides our results to a wide audience, it ensures that our work meets high scientific standards, and we believe that it is a necessary obligation of a research project. The Bibliography of Published Journal Articles section provides citations for peer-reviewed papers co-authored by personnel of project 199102900 that were published from 1998 to 2001.

## **Acknowledgments**

We thank our colleagues at the Bonneville Power Administration, Environmental Protection Agency, Fish Passage Center, Idaho Fish and Game, Idaho Power Company, National Marine Fisheries Service, Nez Perce Tribe, Oregon Department of Fish and Wildlife, Pacific States Marine Fisheries Commission, U. S. Army Corps of Engineers, U. S. Fish and Wildlife Service, U. S. Geological Survey, University of Idaho, and Washington Department of Fish and Wildlife.

## Annual Reporting



## **CHAPTER ONE**

Data and Analyses on Juvenile Snake River  
Fall Chinook Salmon, 1992-2001

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*Abstract.*— In this chapter, I summarize in-season and post-season data and analyses as follows. Early life history timing and growth in 2001 is described and compared to other years. Passage forecast performance in 1999, 2000, and 2001 is assessed. The factors affecting survival for the years 1998–2001 are identified. Fry emergence during 2001 occurred earlier in the upper reach than in the lower reach of the Snake River. Fry emergence in 2001 in the upper reach was similar to other years (2001 median 22 April; 1995–2000 grand mean 22 April). Fry emergence in the lower reach was a little later than during other years (2001 median 6 May; 1992–2000 grand mean 2 May). Shoreline rearing by parr during 2001 occurred earlier in the upper reach than in the lower reach of the Snake River. Rearing timing in the upper reach of the Snake River in 2001 was earlier than during other years (2001 median 29 April; 1995–2000 grand mean 17 May). Rearing timing in the lower reach of the Snake River was also earlier than during other years (2001 median 20 May; 1992–2000 grand mean 30 May). Mean growth rate was higher for parr in the upper reach than in the lower reach of the Snake River. Mean growth rate for parr in the upper reach was slightly lower than during other years (2001 mean 1.1 mm/d; 1995–2000 grand mean 1.2 mm/d). Mean growth rate of parr in the lower reach was the same as during other years (2001 mean 0.9 mm/d; 1992–2000 grand mean 0.9 mm/d). Passage of smolts from the upper reach was later than normal (2001 median 15 July; 1995–2000 grand mean 5 July). Of the 1,163 parr PIT tagged in the lower reach, a total of 185 were detected as smolts as they passed Lower Granite Dam. The median date of passage for fish from the lower reach was earlier than normal (2001 median 7 July; 1992–2000 grand mean 15 July). Passage forecast performance varied among years. The forecasts in 1999 and 2000 performed relatively well except during relatively short time periods in the middle of the run. To the contrary, forecast performance was poor in 2001 because the forecasting method was not developed for such low flow conditions. Mean survival to the tailrace of Lower Granite Dam for the four cohorts in 2001 was only 20.5%, which is the lowest observed for the 1998–2001 time period. Survival was affected by both flow and temperature. The regression model fit to describe survival was: Cohort survival =  $146.94900 + 0.02885 \times \text{Flow} - 7.4998 \times \text{Temperature}$  ( $N = 16$ ;  $P \leq 0.0001$ ;  $R^2 = 0.896$ ). Cohort survival generally increased as flow increased, and decreased as water temperature increased.

## Introduction

Wild subyearling Snake River fall chinook salmon smolts listed for protection under the Endangered Species Act (NMFS 1992) typically migrate seaward in the lower Snake River during late spring and summer when flow is low (chapters four and six in this report). During recovery planning, it was determined that summer water conditions in Lower Granite Reservoir (Figure 1) were unfavorable for smolt survival (NMFS 1995). In July of 1992, a small volume of stored reservoir water was released to increase flow and decrease water temperature in Lower Granite Reservoir. Thereafter, larger volumes of water were released annually between 21 June and 31 August. Releasing this stored water is called summer flow augmentation.

There is not enough reservoir water available to optimize passage conditions in Lower Granite Reservoir throughout the 21 June to 31 August time period. Therefore, interagency management teams (e.g., FPAC and TMT) meet regularly from 21 June to 31 August to apportion the limited water for summer flow augmentation. Fishery managers use their professional judgment and the best available information to develop a water management plan to guide the implementation of summer flow augmentation. This information includes projections of water availability, in-season flow and water temperature data, historic and in-season passage data, and forecasts of smolt passage at Lower Granite Dam (e.g., Connor et al. 2000).

Since 1992, personnel of project 199102900 have been assisting fishery managers during planning, implementation, monitoring, and evaluation of summer flow augmentation. Much of the data provided to fishery managers are in the form of records compiled on wild fall chinook salmon juveniles that were tagged with Passive Integrated Transponder (PIT) tags (Prentice et al. 1990b). The PIT-tag data are provided weekly each year via the PIT-tag Information System, where in turn, it is downloaded, analyzed, and posted in real time on Internet services (e.g., DART; Program RealTime, Burgess and Skalski 2001) to allow managers to track the progress of the smolt migration.

In 2001, I provided members of FPAC and TMT an in-season briefing in early July, 2001 that included: 1) a summary of 2001 catch and tagging data; 2) a comparison of observed time of fry and parr presence among the years 1992 to 2001; 3) a comparison of parr growth among the years 1992 to 2001; and 4) a

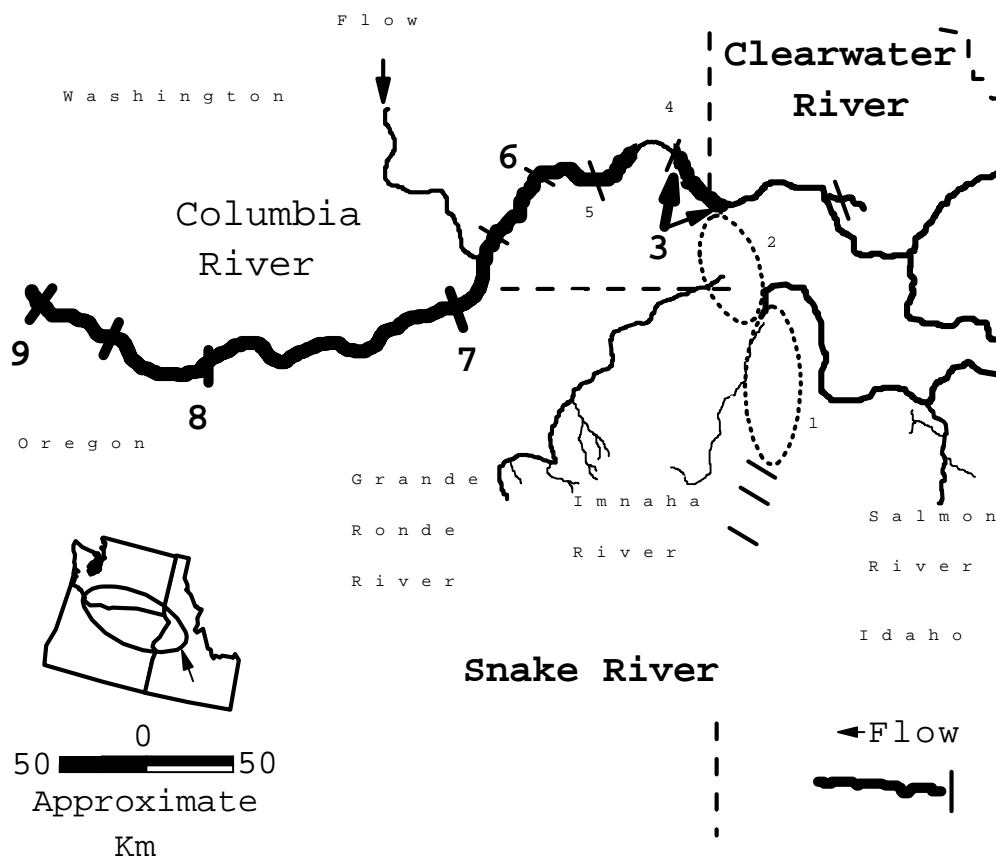


Figure 1—Locations of the upper and lower reaches of the Snake River where adult fall chinook salmon spawn and their offspring were captured by using a beach seine (cross hatched ellipses) and dams where PIT-tagged smolts were detected during seaward migration. The locations are as follows: 1 = Snake River upper reach; 2 = Snake River lower reach; 3 = Lower Granite Reservoir; 4 = Lower Granite Dam; 5 = Little Goose Dam; 6 = Lower Monumental Dam; 7 = McNary Dam; 8 = John Day Dam; and 9 = Bonneville Dam.

forecast of passage at Lower Granite Dam (Figure 1) for wild fall chinook salmon PIT tagged in the Snake River.

I provided the TMT with a post-season briefing in late October, 2001 that included: 1) a comparison of the observed and forecasted passage of wild PIT-tagged subyearling chinook salmon at Lower Granite Dam in 2001; and 2) survival estimates to the tailrace of Lower Granite Dam for wild fall chinook salmon that were PIT-tagged and released upstream of Lower Granite Reservoir in the Snake River.

In this chapter, I summarize in-season and post-season data and analyses as follows. Early life history timing and growth in 2001 is described and compared to other years. Passage forecast performance in 1999, 2000, and 2001 is assessed. The factors affecting survival for the years 1998–2001 are identified.

## **Methods**

*Data collection.*—The Snake River can be divided into two reaches based on differences in water temperature (chapter four in this report). The upper reach extends from the Salmon River confluence to Hells Canyon Dam, and the lower reach extends from the upper end of Lower Granite Reservoir to the Salmon River confluence (Figure 1). Juvenile fall chinook salmon were captured in these two reaches by using a beach seine (Connor et al. 1998). Sampling began in the upstream reach in 1995, and in the lower reach in 1992. Beach seining typically started in April soon after fry began emerging from the gravel, and was conducted weekly at permanent stations within each reach. From 1992 to 1999, additional non-permanent stations were sampled for three consecutive weeks once a majority of fish were at least 60 mm fork length. Only permanent stations were sampled in 2000 and 2001. We discontinued all sampling in June or July when the majority of fish had moved into Lower Granite Reservoir or to points further downstream.

Field personnel inserted PIT tags into parr 60 mm fork length and longer (Connor et al. 1998). Tagged parr were released at the collection site after a 15-min recovery period. Some of the PIT-tagged fish were detected as smolts as they passed downstream in the juvenile bypass systems of dams equipped with PIT-tag monitors (Matthews et al. 1977; Prentice et al. 1990a; Figure 1). Operation schedules for the fish bypass systems varied by dam and year. Most of the detections were in the fish bypass systems of Lower Granite, Little Goose, and Lower Monumental dams operated from early April to early November, and at McNary Dam (Figure 1) operated from early April to early December.

Starting in 1999, non-fin clipped hatchery subyearling fall chinook salmon were released into the Snake River by personnel of the Nez Perce Tribe to supplement wild production. My field staff subjectively identified these fish in the beach seine catch based on body morphology. Hatchery fish were more slender and appeared to have larger eyes than wild fish. Fish judged to be of hatchery-origin were released back to the river and were not included in my analyses.

*Accuracy of origin classification.*—I used two methods to assess how well field personnel identified fish origin (i.e., hatchery versus wild). In 2000, known hatchery fish that had been coded-wire tagged prior to release in the Snake River (no wild fish were coded-wire tagged) were used as a baseline. The origin of every fish captured was judged, and then the fish was passed through a coded-wire tag detector. I calculated classification accuracy as the number of fish judged to be hatchery fish, divided by the number of coded-wire tagged fish collected, multiplied by 100.

In 2001, classification accuracy was based on fish captured during beach seining that had wounds or scars resulting from previous PIT tagging (hereafter, recaptured fish). Every fish was examined for wounds and scars after removing it from the seine. The origin of each recaptured PIT-tagged fish was judged, and then the fish was scanned for its unique tag code. I traced the origin of each recaptured fish based on this unique tag code. I calculated both within-origin (hatchery and wild, separately) and across-origin (hatchery and wild, combined) classification accuracy. Within-origin classification accuracy was the number of correct classifications divided by the number of recaptured hatchery or wild fish. Across-origin classification accuracy was calculated as the weighted average of the within-run classification accuracy estimates.

*Early life history and growth.*—I used the capture dates of wild fall chinook salmon smaller than 45-mm fork length to describe time of presence for newly emergent fry (hereafter emergence timing). I used the capture dates for fish 45-mm fork length and longer to describe time of presence for fall chinook salmon parr (hereafter, rearing timing). All capture dates were adjusted to Sunday's date the week of sampling to account for differences in day of sampling between the upper and lower reaches of the Snake River. For example, a capture date of 2 May, 1993 (Sunday) was reported for fry and parr collected from 4 May to 6 May (Tuesday to Thursday).

Absolute growth rate (mm/d) during shoreline rearing was calculated using length data from PIT-tagged parr recaptured by beach seine after initial capture and tagging. Absolute growth rate was calculated as: fork length at recapture minus fork length at initial capture divided by the number of days between initial capture and recapture.

I used the PIT-tag detection data at Lower Granite Dam to represent the onset of seaward migration by subyearling smolts.

*Passage forecasting.*—I applied a passage forecasting method (Connor et al. 2000) to in-season data collected in 1999, 2000 and 2001. This method uses a model to predict which PIT-tagged fish will survive to pass Lower Granite Dam, and a second model to predict when these survivors will pass the dam. The forecast is in the form of a cumulative passage distribution plus and minus a 90% forecast interval.

I assessed the 1999, 2000, and 2001 passage forecasts two ways. First, I determined the difference in days between forecasted and observed dates when 50, 70, 80, and 90% passage was complete. Second, I determined how many days the forecast and its 90% intervals did not contain observed passage. Forecast performance was considered to be poor when observed was not contained by these intervals.

*Survival.*—The first step in the analysis was to divide the annual samples of PIT-tagged fall chinook salmon into four sequential within-year release groups referred to as "cohorts." I divided the annual samples into cohorts based on estimated fry emergence dates. I estimated fry emergence date for each fish in two steps. First, the number of days since each PIT-tagged fish emerged from the gravel was calculated by subtracting 36 mm from its fork length measured at initial capture, and then dividing by the daily growth rate observed for recaptured PIT-tagged fish within each reach. The 36-mm fork length for newly emergent fry was the mean of the observed minimum fork lengths. Second, emergence date was estimated for each fish by subtracting the estimated number of days since emergence from its date of initial capture, tagging, and release. I sorted the data in ascending order by estimated fry emergence date, and then divided it into four cohorts of approximately equal numbers of fish.

The single release-recapture model (Cormack 1964; Skalski et al. 1998) was used to estimate survival probability to the tailrace of Lower Granite Dam for each cohort. I insured that the single release-recapture model fit the data by using three assumption tests described by Burnham et al. (1987) and Skalski et al. (1998). I refer the reader to Chapter Six in this report

for a detailed description of the statistical methods used to describe the factors affecting survival.

## Results

### *Accuracy of Origin Classification*

In 2000, 57 of the fall chinook salmon captured during beach seining were coded-wire tagged. Of these 55 were judged to be of hatchery origin, which equates to a within-origin classification accuracy of 96.5% (Table 1). In 2001, we recaptured 218 previously PIT-tagged fish. Within-origin classification accuracy was 92.7% for hatchery fall chinook salmon and 99.4% for wild fall chinook salmon, which equates to an across-origin classification accuracy of 97.7% (Table 1).

Table 1.—Within- and across-origin classification accuracies (%). Classification of origin (hatchery vs. wild) was based on a subjective assessment of body morphology, and then verified based on coded-wire (year 2000) or PIT tags (year 2001).

Actual origin	n	Number classified into each origin		Classification accuracy	
		Hatchery	Wild	Within	Across
Year 2000					
Hatchery	57	55	2	96.5	N/A
Year 2001					
Hatchery	55	52	4	92.7	98.2
Wild	163	1	162	97.7	

### *Early Life History and Growth*

In 2001, a total of 7,004 wild fall chinook salmon was captured by beach seine (including marked fish captured more than once). Of these, 2,624 were fry. Fry emergence during 2001 occurred earlier in the upper reach than in the lower reach of the Snake River based on time of fry presence (Table 2). Fry emergence in 2001 in the upper reach was similar to other years (2001 median 22 April; 1995–2000 grand mean 22 April). Fry emergence in the lower reach was a little later than during



other years (2001 median 6 May; 1992–2000 grand mean 2 May).

A total of 4,380 of the wild fall chinook salmon captured in 2001 were parr. Shoreline rearing by parr during 2001 occurred earlier in the upper reach than in the lower reach of the Snake River (Table 3). Rearing timing in the upper reach of the Snake River in 2001 was earlier than during other years (2001 median 29 April; 1995–2000 grand mean 17 May). Rearing timing in the lower reach of the Snake River was also earlier than during other years (2001 median 20 May; 1992–2000 grand mean 30 May).

A total of 1,392 of the wild fall chinook salmon parr was PIT tagged in 2001. Of these, 137 were recaptured during beach seining. Mean growth rate was higher for parr in the upper reach than in the lower reach of the Snake River (Table 4). Mean growth rate for parr in the upper reach was slightly lower than during other years (2001 mean 1.1 mm/d; 1995–2000 grand mean 1.2 mm/d). Mean growth rate of parr in the lower reach was the same as during other years (2001 mean 0.9 mm/d; 1992–2000 grand mean 0.9 mm/d).

A total of 13 of the 229 parr PIT tagged in the upper reach of the Snake River was detected as smolts as they passed Lower Granite Dam. Passage of smolts from the upper reach was later than normal (2001 median 15 July; 1995–2000 grand mean 5 July). Of the 1,163 parr PIT tagged in the lower reach, a total of 185 were detected as smolts as they passed Lower Granite Dam. The median date of passage for fish from the lower reach was earlier than normal (2001 median 7 July; 1992–2000 grand mean 15 July). For the first time on record, fish tagged in the lower reach passed Lower Granite Dam earlier than fish tagged in the upper reach (Table 5).

Table 2.—Emergence timing (given as Sunday's date for each week) of wild fall chinook salmon fry in the upper and lower reaches of the Snake River, 1992 to 2001. The range of dates is given in parentheses.

Year	N	Median dates of presence
<b>Snake River upper reach</b>		
1995	117	23 Apr (02 Apr to 21 May)
1996	14	28 Apr (14 Apr to 05 May)
1997	1	20 Apr (N/A)
1998	101	19 Apr (12 Apr to 10 May)
1999	97	02 May (04 Apr to 23 May)
2000	683	09 Apr (02 Apr to 14 May)
2001	1,356	22 Apr (01 Apr to 20 May)
<b>Snake River lower reach</b>		
1992	355	26 Apr (29 Mar to 24 May)
1993	199	16 May (04 Apr to 20 Jun)
1994	440	15 May (03 Apr to 05 Jun)
1995	257	30 Apr (02 Apr to 04 Jun)
1996	268	05 May (14 Apr to 23 Jun)
1997	114	04 May (20 Apr to 29 Jun)
1998	322	26 Apr (12 Apr to 14 Jun)
1999	278	02 May (04 Apr to 27 Jun)
2000	415	09 Apr (02 Apr to 04 Jun)
2001	1,268	06 May (01 Apr to 03 Jun)

Table 3.—Rearing timing (given as Sunday's date for each week) of wild fall chinook salmon parr in the upper and lower reaches of the Snake River, 1992 to 2001. The range of dates is given in parentheses.

Year	N	Median dates of presence
<b>Snake River upper reach</b>		
1995	985	28 May (09 Apr to 18 Jun)
1996	118	12 May (14 Apr to 16 Jun)
1997	119	25 May (20 Apr to 15 Jun)
1998	1,078	17 May (12 Apr to 05 Jul)
1999	1,493	23 May (11 Apr to 27 Jun)
2000	1,064	23 Apr (02 Apr to 11 Jun)
2001	1,303	29 Apr (01 Apr to 10 Jun)
<b>Snake River lower reach</b>		
1992	1,765	17 May (29 Mar to 07 Jun)
1993	2,215	06 Jun (11 Apr to 18 Jul)
1994	4,346	29 May (03 Apr to 10 Jul)
1995	1,408	04 Jun (02 Apr to 02 Jul)
1996	756	26 May (14 Apr to 14 Jul)
1997	938	08 Jun (20 Apr to 13 Jul)
1998	2,512	31 May (12 Apr to 05 Jul)
1999	1,647	06 Jun (04 Apr to 11 Jul)
2000	1,578	14 May (02 Apr to 25 Jun)
2001	3,078	20 May (01 Apr to 17 Jun)

Table 4.—Mean growth rates (mm/d+SD) for wild fall chinook salmon parr collected in the upper and lower reaches of the Snake River, 1992 to 2001. Sample sizes are given in parentheses

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Growth rate by reach		
Year	Upper reach	Lower reach
1992		0.9+0.1 ( 66)
1993		0.7+0.4 (202)
1994		1.1+0.4 (341)
1995	1.2+0.3 (148)	1.0+0.4 ( 78)
1996	1.1+0.3 ( 19)	0.9+0.4 ( 49)
1997	1.3+0.3 ( 20)	0.8+0.3 ( 80)
1998	1.1+0.3 (112)	0.9+0.3 (129)
1999	1.3+0.3 (171)	1.1+0.3 ( 92)
2000	1.3+0.2 ( 90)	1.0+0.3 ( 40)
2001	1.1+0.1 ( 15)	0.9+0.2 (122)

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Table 5.—Smolt migration timing at Lower Granite Dam for wild fall chinook salmon that were initially captured, PIT tagged, and released in the upper and lower reaches of the Snake River, 1992–2001. The range of dates is given in parentheses.

Year	N	Median dates of detection
<b>Snake River upper reach</b>		
1995	203	18 Jul (04 Jun to 24 Oct)
1996	19	04 Jul (20 May to 25 Jul)
1997	22	27 Jun (04 Jun to 13 Aug)
1998	173	07 Jul (19 May to 21 Aug)
1999	319	03 Jul (02 Jun to 28 Aug)
2000	72	27 Jun (06 May to 18 Jul)
2001	13	15 Jul (24 Jun to 29 Oct)
<b>Snake River lower reach</b>		
1992	39	20 Jun (04 May to 21 Jul)
1993	234	21 Jul (31 May to 25 Oct)
1994	193	17 Jul (23 May to 01 Nov)
1995	238	01 Aug (02 Jun to 26 Oct)
1996	126	22 Jul (17 May to 31 Oct)
1997	97	16 Jul (14 Jun to 13 Oct)
1998	380	11 Jul (29 May to 19 Oct)
1999	241	25 Jul (01 Jun to 30 Aug)
2000	257	02 Jul (18 May to 28 Oct)
2001	185	07 Jul (16 May to 26 Oct)

### *Passage Forecasts*

Passage at Lower Granite Dam in 1999 by wild fall chinook salmon smolts from the Snake River was forecasted to be 50% complete by 8 July, 70% complete by 14 July, 80% complete by 25 July, and 90% complete by 11 August (Figure 2). The 1999 forecast predicted 50% passage at Lower Granite Dam eight days earlier than was observed, 70% passage two days earlier than was observed, 80% passage five days earlier than was observed, and 90% passage four days later than was observed. The 1999 forecast predicted passage poorly from 11 July to 20 July (Figure 2). When viewed across the entire period of passage as designed, the 90% forecast intervals contained observed passage 94% of the time.

Passage at Lower Granite Dam in 2000 by wild fall chinook salmon smolts from the Snake River was forecasted to be 50% complete by 8 July, 70% complete by 11 July, 80% complete by 14 July, and 90% complete by 19 July (Figure 2). The 2000 forecast predicted 50% passage at Lower Granite Dam 7 days later than was observed, 70% passage 1 day later than was observed, 80% passage 3 days earlier than was observed, and 90% passage 18 days earlier than was observed. The 2000 forecast predicted passage poorly from 1-July to 7-July (Figure 2). When viewed across the entire period of passage as designed, the 90% forecast intervals contained observed passage 96% of the time.

Passage at Lower Granite Dam in 2001 by wild fall chinook salmon smolts from the Snake River was forecasted to be 50% complete by 8 July, 70% complete by 11 July, 80% complete by 11 July, and 90% complete by 14 August (Figure 2). The 2001 forecast predicted 50% passage at Lower Granite Dam 10 days later than was observed, 70% passage 14 days later than was observed, 80% passage 19 days later than was observed, and 90% passage on about the day it was observed. The 2001 forecast predicted passage poorly from 3-July to 23-July (Figure 2). When viewed across the entire period of passage as designed, the 90% forecast intervals contained observed passage 88% of the time.

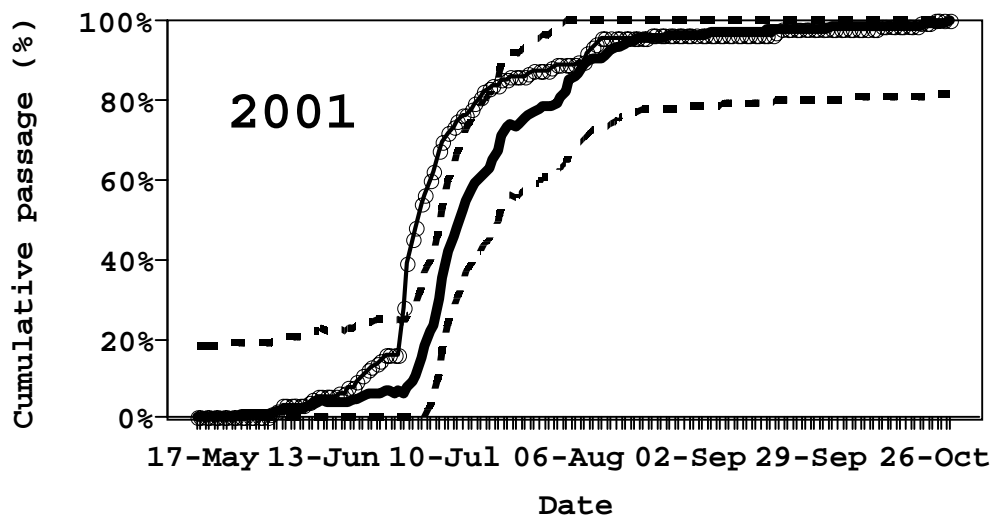
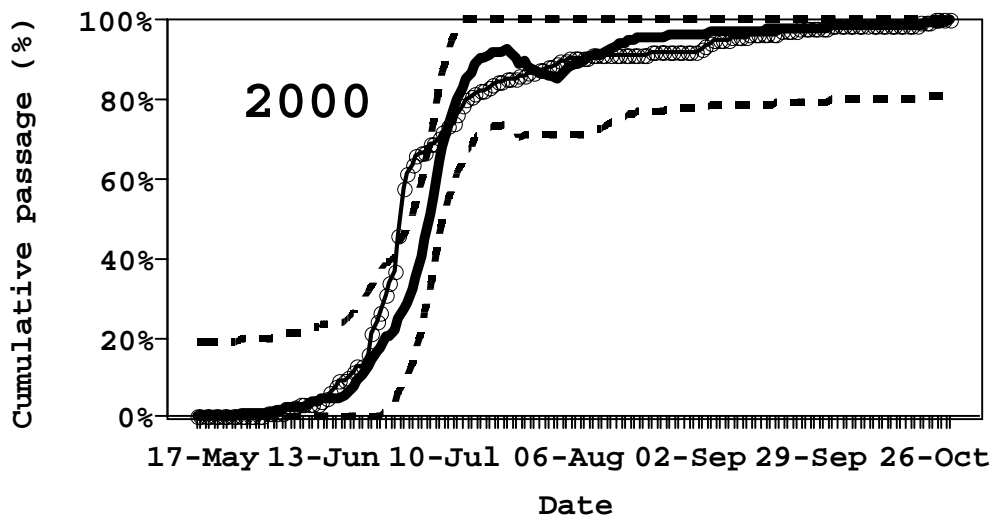
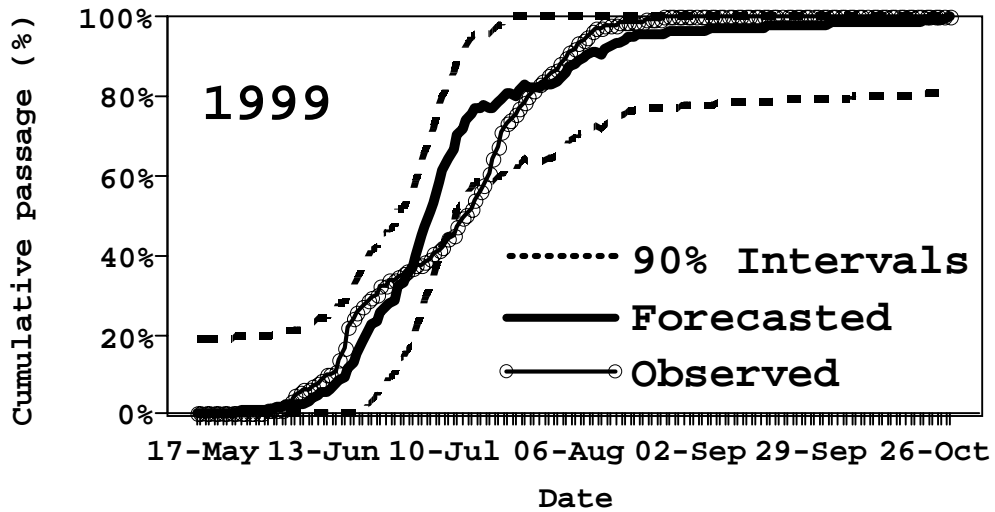


Figure 2.—Forecasted versus observed cumulative passage (%) at Lower Granite Dam for wild fall chinook salmon PIT tagged in the Snake River, 1999-2001.

## Survival

Mean survival to the tailrace of Lower Granite Dam for the four cohorts in 2001 was only 20.5%, which is the lowest observed for the 1998-2001 time period (Table 6). Survival for cohort 1 (the earliest migrating fish with the highest survival) was a total of 23.8 percentage points below the 1998-2001 average for cohort 1. Survival for cohorts 2 was 33.8 and 31.7 percentage points below the 1998-2001 averages for these two cohorts (Table 6). Survival for cohort 4 (the latest migrating fish with the lowest survival) was 23.9 percentage points lower than the 1998-2001 average for this cohort.

The values for the predictor variables assessed as factors affecting survival are given in Table 7. Fork length and flow ( $N = 16$ ;  $r = 0.64$ ;  $P = 0.008$ ) and fork length and temperature ( $N = 16$ ;  $r = -0.64$ ;  $P = 0.008$ ) were collinear. Therefore, fork length was not entered with flow and temperature into the same multiple regression model. Fork length and release date ( $N = 16$ ;  $r = 0.50$ ;  $P = 0.05$ ), release date and flow ( $N = 16$ ;  $r = -0.04$ ;  $P = 0.88$ ), release date and temperature ( $N = 16$ ;  $r = 0.33$ ;  $P = 0.21$ ), and flow and temperature ( $N = 16$ ;  $r = -0.59$ ;  $P = 0.02$ ) were non-collinear based on the criterion of  $r = 0.6$ .

The slope coefficient for the variable release date never differed significantly ( $\alpha = 0.05$ ) from zero when entered into models including fork length, flow, and temperature. The model that predicted cohort survival from flow and temperature had a Mallows'  $C_p$  score one less than the number of parameters, the lowest AIC value, and an  $R^2$  of 0.896 (Table 8). The next best models were bivariate and included the variable flow ( $P = \leq 0.0001$ ), or temperature ( $P = 0.0008$ ), or fl ( $P = 0.0008$ ) (Table 8).

The final (i.e., the best) multiple regression model was: Cohort survival =  $146.94900 + 0.02885 \times \text{Flow} - 7.4998 \times \text{Temperature}$ . The final model was significant ( $N = 16$ ;  $P \leq 0.0001$ ) as were the coefficients for flow ( $t = 6.43$ ;  $P \leq 0.0001$ ) and temperature ( $t = -2.95$ ;  $P = 0.011$ ). Flow and temperature explained 89.6% of the observed variability in cohort survival to the tailrace of Lower Granite Dam. Cohort survival generally increased as flow increased, and decreased as water temperature increased.



Table 6.—Estimates of survival probability (% $\pm$ SE) to the tailrace of Lower Granite Dam for cohorts of wild subyearling fall chinook salmon, 1998 to 2001.

Cohort	Survival by year				Cohort means
	1998	1999	2000	2001	
1	70.8 $\pm$ 2.9	87.7 $\pm$ 4.6	57.1 $\pm$ 4.1	40.1 $\pm$ 3.1	63.9
2	66.1 $\pm$ 3.3	77.0 $\pm$ 3.8	53.4 $\pm$ 4.2	20.5 $\pm$ 2.5	54.3
3	52.8 $\pm$ 3.1	81.2 $\pm$ 5.8	44.4 $\pm$ 3.6	17.2 $\pm$ 3.0	48.9
4	35.6 $\pm$ 2.9	36.4 $\pm$ 3.5	35.7 $\pm$ 4.3	4.0 $\pm$ 1.3	27.9
Annual means	56.3	70.6	47.7	20.5	

Table 7.—Predictor variables assessed as factors affecting survival to the tailrace of Lower Granite Dam for each cohort of wild subyearling fall chinook salmon, 1998–2001. Abbreviations: Date = median day of year of release; Fl = mean fork length (mm) at release; Flow = a flow (m<sup>3</sup>/s) exposure index calculated as the mean flow measured at Lower Granite Dam during the period when the majority of smolts passed the dam; and, Degrees = a water temperature (°C) exposure index calculated as the mean temperature measured in the tailrace of Lower Granite Dam during the period when the majority of smolts passed the dam. See Chapter Six methods for exposure index calculations.

Cohort	N	Date	Fl	Flow	Degrees
<b>Year 1998</b>					
1	515	140	80	2,344	17.6
2	515	141	75	2,021	18.7
3	515	153	73	1,898	19.0
4	515	167	70	1,299	19.8
<b>Year 1999</b>					
1	441	147	80	2,378	16.3
2	440	153	77	1,963	17.1
3	440	152	70	2,116	16.7
4	440	167	68	1,353	18.3
<b>Year 2000</b>					
1	303	130	77	1,510	16.7
2	302	144	77	1,296	17.6
3	302	146	77	1,274	17.8
4	302	158	71	859	18.5
<b>Year 2001</b>					
1	348	135	74	754	18.5
2	348	142	69	743	18.9
3	348	143	68	753	19.2
4	348	155	66	746	18.8

Table 8.—Mallow's Cp scores, Akaikes information criteria (AIC), and coefficients of determination ( $R^2$ ) used to compare the fit of multiple regression models describing the survival of cohorts of wild subyearling fall chinook salmon from release in the Snake River to the tailrace of Lower Granite Dam, 1998 to 2001. Abbreviations: Date = median day of year of release; Fl = mean fork length (mm) at release; Flow = a flow ( $\text{m}^3/\text{s}$ ) exposure index calculated as the mean flow measured at Lower Granite Dam during the period when the majority of smolts passed the dam; and, Degrees = a water temperature ( $^{\circ}\text{C}$ ) exposure index calculated as the mean temperature measured in the tailrace of Lower Granite Dam during the period when the majority of smolts passed the dam.

C(p)	AIC	$R^2$	Variables in model
2.3	70.4	0.896	Flow Degrees
8.6	76.7	0.827	Flow
39.5	91.3	0.566	Degrees
2.6	91.4	0.563	Fl

## Discussion

### *Life History, Growth, Passage Forecasts, and Survival*

Fall chinook salmon fry emergence in the upper and lower reaches of the Snake River occurred over the "typical" time period in 2001. Emergence is expected to be earlier in the upper reach than in the lower reach because winter-spring temperatures are warmer in the upper reach (see Chapter Four in this report). Shoreline rearing also occurred earlier in the upper reach than in the lower reach, and rearing in both reaches was earlier than normal. Passage of smolts at Lower Granite Dam was later than normal for fish tagged in the upper reach, whereas passage was earlier than normal for fish tagged in the lower reach. Consequently, for the first time on record fish tagged in the lower reach passed Lower Granite Dam before fish tagged in the upper reach. The operation of Hells Canyon Complex of dams during spring 2001 offers one plausible explanation for this discrepancy. Dramatic changes in stream discharge resulting from hydropower operations displaced many young fall chinook salmon from the upper reach before they were large enough to tag, thus the rearing period was truncated. These fish were then tagged as they reared along the shoreline of the lower reach, which resulted in what appeared to be early rearing and seaward migration by fish of lower reach-origin.

Parr in the upper reach grew faster than those in the lower reach. This is expected because the upper reach is warmer during the period young fall chinook salmon are rearing along the shoreline. Parr grew rapidly in both reaches. Management activities with the potential to decrease growth rates below 1995-2000 levels should be monitored. Releasing large numbers of hatchery fall chinook salmon into the Snake River to supplement wild production might eventually reduce growth through intra-specific competition. Growth rate could be used to index the effects of the supplementation on the well-being of wild fall chinook salmon in the Snake River. For a more detailed analysis and discussion of growth, I refer the reader to Chapter Seven in this report.

I found that passage forecast performance varied among years. The forecasts in 1999 and 2000 performed relatively well except during relatively short time periods in the middle of the run. To the contrary, forecast performance was poor in 2001 because the forecasting method was not developed for such low flow conditions (see Connor et al. 2000). Two patterns can be seen in the 1999-2001 forecasts. First, the forecast predicted

earlier passage than was observed in 1999 when environmental conditions for survival were favorable. Second, the forecast predicted later passage than was observed in 2000 and 2001 when environmental conditions were unfavorable for survival. More accurate forecasts might be possible if separate survival and passage models were fit for application in low and high flow years.

Survival of fall chinook salmon smolts in 2001 was low. Fall chinook salmon survival was also low during the low flow years of 1992 and 1994. The average flow between 21 June and 31 August in 1992 was only 539 m<sup>3</sup>/s and the maximum summer temperature was 23.9°C, whereas in 1994 flow average 744 m<sup>3</sup>/s and maximum summer temperature was 19.9°C (Connor et al. 1998). Technical and logistical limitations prevented me from using the single release-recapture model to estimate survival during 1992 and 1994. In the simplest sense, the single release-recapture model calculates survival by dividing detection rate by fish guidance efficiency. Detection rates for PIT-tagged wild fall chinook salmon at Lower Granite Dam in 1992 and 1994 were 5.1% and 8.4%, respectively. These detection rates equate to survivals of 16% and 25% assuming a fish guidance efficiency of 33%. Relatively low survival of fall chinook salmon during low flow years and the relatively high survival in average to high flow years (e.g., 71% in 1999) strengthens the argument for a flow-temperature-survival relation.

The regression results in this chapter, and in Chapter Six in this report, suggest that survival of fall chinook salmon smolts to the tailrace of Lower Granite Dam is directly related to flow and inversely related to temperature. I was able to enter flow and temperature into the same 1998-2001 regression model because the collinearity between the two variables was relatively weak. Entering both variables in the same model might not be feasible in future years because flow and temperature are typically correlated. The regression model suggests that many smolts did not survive to pass Lower Granite Dam in 2001 because flow in Lower Granite Reservoir was critically low, thus smolts spent extended periods of time in relatively warm water. I refer the reader to Chapters Five and Six in this report for an in-depth discussion of flow and temperature conditions provided by summer flow augmentation that might work together to affect downstream migration rate and survival of young fall chinook salmon in Lower Granite Reservoir.

### *The Future Direction of Project 199102900*

Summer flow augmentation will continue to be implemented in 2002 based on results of past studies and reviews (e.g., ISAB 2001). In-season planning and post-season monitoring and evaluation should continue. Fishery managers need more information on: (1) the effect of summer flow augmentation on velocity and temperature in Lower Granite Reservoir; (2) the roles velocity and temperature play on migratory behavior of juvenile fall chinook salmon in Lower Granite Reservoir; and (3) where and when juvenile fall chinook salmon die during freshwater rearing and early seaward migration.

Beginning in 2002, the objectives of project 199102900 will be to: (1) provide information to fishery managers to maximize the effectiveness of summer flow augmentation; and (2) provide a better understanding of how summer flow augmentation affects water temperature, water velocity, juvenile fall chinook salmon migratory behavior, and juvenile fall chinook survival salmon in Lower Granite Reservoir. By accomplishing these two objectives, personnel of project 199102900 will continue to help fishery managers recover fall chinook salmon in the Snake River basin.

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## CHAPTER TWO

Survival Estimates of PIT-tagged subyearling fall chinook salmon  
released in the Hanford Reach and McNary Reservoir of the  
Columbia River, 2001

by

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## Introduction

The naturally spawning stock of fall chinook salmon (*Oncorhynchus tshawytscha*) in the Hanford Reach may be the furthest inland healthy fall chinook salmon population that exists in the Columbia River Basin (Huntington et al. 1996). The Hanford Reach is the last remaining unimpounded stretch of the Columbia River above Bonneville Dam in the United States. Although upstream dams control flows in Hanford Reach, the river shorelines are relatively undeveloped. Juvenile fall chinook salmon naturally spawned in the Hanford Reach rear in areas near the shoreline (Becker 1973), and within a few months migrate seaward through the riverine reach and a series of four dams and their reservoirs before reaching the estuary.

The success of fall chinook salmon in the Hanford Reach compared to the ESA-listed stock of naturally spawning fall chinook salmon in the Snake River is not fully understood. The fewer number of dams and reservoirs that must be passed by downstream migrating juveniles and by upstream migrating adults is one possible reason. In addition, the Columbia River typically has higher flows and lower water temperature than does the Snake River, which may contribute to increased survival of the Hanford Reach stock. Because rearing habitats are somewhat similar in both rivers and both fall chinook salmon populations have the same life history, understanding the survival of Hanford Reach fall chinook salmon may provide an understanding of what limits the Snake River population.

Survival estimation for migratory fish requires at least two downstream detection sites. Because of the PIT-tag interrogation infrastructure in the Snake River, survival estimates have been calculated for both hatchery and natural subyearlings from the Hells Canyon since the early 1990s (Connor et al. 1998; Muir et al. 1998; Smith et al. 2002; Connor et al. Chapter One this report). PIT-tag interrogation systems are operated at McNary, John Day, and Bonneville dams, which make it possible to estimate survival of juvenile Hanford Reach fall chinook salmon to below John Day Dam.

A study conducted by the National Marine Fisheries Service has estimated survival of migratory juvenile fall chinook salmon below McNary Dam (Smith et al. 2002). In this study, we estimated the reach specific survival for subyearlings released in the Hanford Reach and McNary Reservoir. Information from our study combined with information derived from other survival

studies may increase managers' understanding as to why the Hanford population has remained a viable stock in spite of the modified environment encountered during migration to the estuary and how it may be further enhanced. Our study objectives were to: 1) estimate the survival of subyearling fall chinook salmon released in the Hanford Reach and McNary Reservoir to McNary Dam, and 2) determine if seasonal differences in survival exist.

## **Study Area**

The Hanford Reach is the only remaining unimpounded section of the Columbia River in the United States above Bonneville Dam. Flows are regulated on an hourly basis by Priest Rapids Dam, and by other main-stem dams upriver. The Hanford Reach extends from Priest Rapids Dam tailrace, river kilometer (rkm) 639.1, to the head of McNary Reservoir (Lake Wallula) at approximately rkm 568 at maximum pool elevation (Figure 1). McNary Reservoir was created by impoundment of the Columbia River behind McNary Dam (rkm 470.7). McNary Dam is the first of the four lower Columbia River dams subyearlings must pass on their seaward migration. John Day Dam (rkm 348.6) and Bonneville Dam (rkm 234.5) also have juvenile bypass systems with PIT-tag interrogation systems. The Dalles Dam (rkm 308.9) is the only dam downstream from the release locations without detection capability. River kilometers are measured from the mouth of the Columbia River.

## **Methods**

### *General Approach*

To estimate the survival of subyearling fall chinook salmon migrating through the Hanford Reach and McNary Reservoir, five releases were made at two different times (an early period and a late period) in both the upper end of the Hanford Reach and in the upper end of McNary Reservoir in 2001. Statistical models were used to calculate the survival from the release site to McNary Dam for each release. The five releases common in space and time are referred to as groups. Information on flow, spill, and water temperature at Columbia River dams were downloaded from the Columbia River Data Access in Real Time database ([www.cbr.washington.edu](http://www.cbr.washington.edu)).

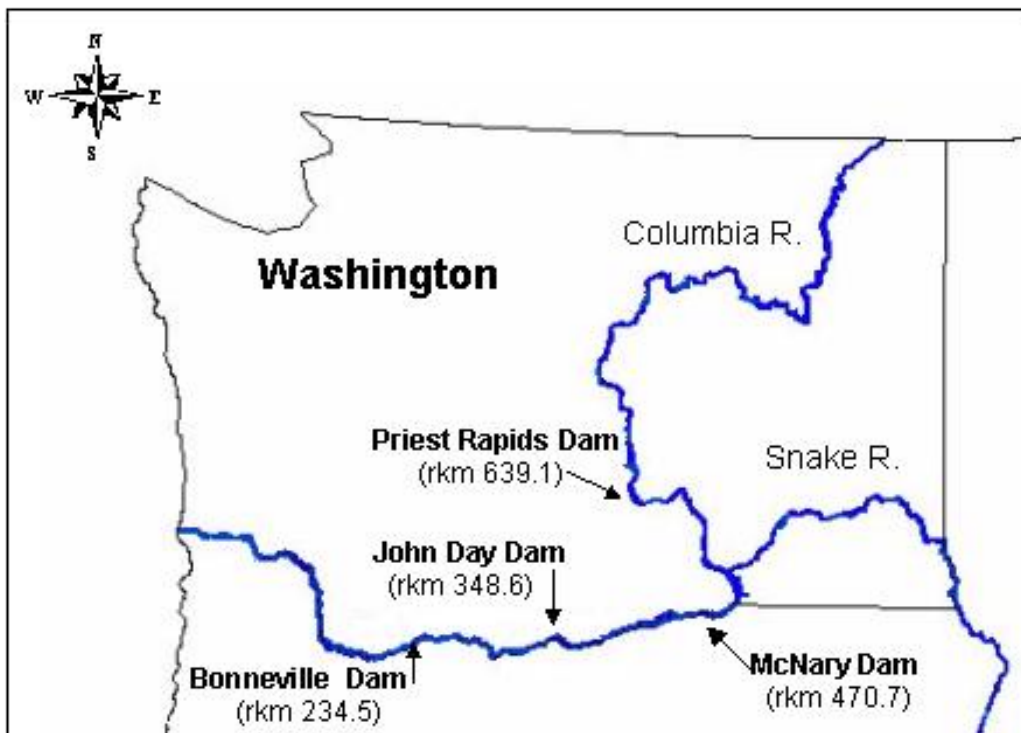


Figure 1.-Map of the Columbia River and the dams downstream of the release locations of PIT-tagged subyearling fall chinook salmon in 2001.

### *Fish Collection*

All subyearlings were collected in the Hanford Reach or at the head of McNary Reservoir. Subyearlings were collected between rkm 541 and 558 for both the riverine and the reservoir releases and tagged near Richland, Washington. Subyearling fall chinook salmon were collected using beach seines in shoreline areas. We limited the number of fish we collected in each seine haul to reduce crowding stress. Fish were transferred to holding containers where they were sorted and subyearlings approximately 62 mm, or larger, were kept for tagging. Fish were not anesthetized or measured during collection and sorting. Fish were held overnight in net pens to allow their gut contents to evacuate and recover from handling stress prior to tagging.

### *Tagging and Release*

Tagging procedures followed those outlined by the Columbia Basin Fish and Wildlife Authority PIT Tag Steering Committee (1999) based on methods and equipment outlined by Prentice et al. (1990). Subyearlings held overnight in net pens were transferred to holding containers receiving a continuous flow of fresh river water the following day. Temperature was continuously monitored at all input and output locations to ensure that subyearlings did not experience temperature fluctuations exceeding that of the normal main-stem river. Aeration was provided where flow-through was not possible, such as in the anesthesia. Subyearlings were anesthetized ten at a time with Tricaine Methansulfonate (MS-222), injected with a PIT-tag, using a 12-gauge hypodermic syringe, and fork length recorded. Prior to tagging, the fish were inspected for adipose clips and previous PIT-tagging scars. Fish with clips were rejected and scarred fish were scanned for possible tag codes and rejected. Each day, the first 50 subyearlings were weighed and measured (fork length), and the remaining fish tagged that day were measured only. Tagged fish were held in a dark container until they recovered, mortalities and shed tags were removed, and tagged fish were transferred to an overnight holding net pen for release the following day. Tags from fish that did not show signs of recovery were removed from the sample and reused after being removed from the database. Subyearlings were held until the following morning to determine post-tagging mortality.

Approximately 1,000 fish were tagged each day in two 5-d tagging sessions (June 7-12 and June 27-July 1). Fish tagged each day were divided into two groups for subsequent paired

release in the Hanford Reach and McNary Reservoir. This resulted in releasing about 2,500 fish over 5 d in both the Hanford Reach and McNary Reservoir in early and late June.

Fish were hauled by truck to release sites in the Hanford Reach (Vernita Bridge at rkm 626) and to McNary Reservoir near Richland, Washington (rkm 543). Fish released in McNary Reservoir were held and driven about in the truck for the same amount of time as fish released further up river, approximately 1.25 h. Fish were hauled in oxygenated water that was maintained to  $\pm 0.5^{\circ}\text{C}$  of ambient river temperature. Fish were generally released in about 10 minutes after arrival at the release site.

### *Survival Estimation*

Tagging and detection information that we used to estimate survival was downloaded from the PIT Tag Information System (PTAGIS). Tag files were reviewed and adjusted where necessary to account for mortalities, removals, tag loss, recaptures, inaccuracies, and inconsistencies. Fish that were recaptured in beach seining activities prior to arrival at McNary Dam were removed from the analysis to minimize bias associated with additional handling. Fish recaptured by Washington Department of Fish and Wildlife coded-wire tagging operations in the Hanford Reach were also removed.

A detection history was constructed for each tagged fish. Because fish mortalities with shed tags that occurred prior to release that could not be accounted for in the tag files, we adjusted for tags shed during post-tagging holding by removing an equal number of tag codes from undetected fish prior to survival estimation. We assumed our detection probability is the probability that a fish migrated as a subyearling and did not include fish that may have migrated as yearlings.

We examined the effect of fish size at release on subsequent detection at McNary Dam. We used a Kolmogorov-Smirnov test for homogeneity of discrete distributions for each release to determine whether a difference in the size distribution existed between detected and undetected fish and the probability that they were drawn from the same distribution. If significant differences were found in four or more releases in a group, then we assumed these differences applied to the group as a whole.

The single release-recapture version of the Cormack/Jolly-Seber survival model (Cormack 1964; Jolly 1965; Seber 1965; Burnham et al. 1987) was used to estimate the proportion of each release group surviving to McNary Dam. This analysis was performed using the computer program SURPH (Survival with Proportional Hazards) (Smith et al. 1994). Chi-square tests of homogeneity and visual plots of passage distributions were used to evaluate whether the groups had similar passage timing. Similar passage timing is important if conditions within the reaches are to be compared. Additional assumptions and tests are discussed in Burnham et al. (1987).

Survival estimates were calculated for individual release groups from release to McNary Dam. A weighted average and standard error was computed for each of the individual groups (Burnham et al. 1987). The weighted averages were compared to determine if the survival estimates and joint survival and detection probabilities ( $\lambda$ ) differed by release location (e.g., Hanford and McNary) within a time period (e.g., early June and late June) and if they differed seasonally at the same release location. We made these comparisons using a two-tailed Z test of equal survival, and assumed statistical significance at  $P < 0.10$ . We also calculated travel times for each tagged fish from Vernita Bridge (Hanford release) to McNary Dam and from Richland, Washington (reservoir release) to McNary Dam. We defined travel time as the number of days from time of release to the first detection in the McNary Dam bypass system.

## Results

Subyearling fall chinook salmon were tagged on two occasions from June 1 to June 12 and from June 27 to July 1, 2001. Mean sizes of fish tagged ranged from 65 mm (SD=3.5 mm) in early June to 72.3 mm (SD=4.0 mm) in late June. Post-tagging mortality averaged 3.6% (SD=2.3%, range=0.2-8.0%). The number of fish released and the number mortalities from tagging are recorded in Table 1.

Subyearlings from all release groups were detected at McNary Dam (Table 2). The percentage of fish detected ranged from 14.7 (late June release in McNary Reservoir) to 30.0 (first early June release in the Hanford Reach). The numbers and percentages of fish detected at McNary Dam were lower for the late June releases in both the Hanford Reach and McNary Reservoir than for the early June releases. We found significant differences between the size distributions of fish

Table 1.-Number of Hanford Reach subyearling fall chinook salmon PIT tagged and released 24 h later, tagging mortalities, and release locations in 2001. Tagging mortalities include all sources of mortality accumulated to 24 h post-tagging.

Release date	Release location	Number tagged	Tagging mortalities	Percent mortalities	Number released
06/07/01	Hanford	350	5	1.4	345
06/07/01	McNary	415	10	2.4	405
06/09/01	Hanford	525	2	0.4	523
06/09/01	McNary	520	1	0.2	519
06/10/01	Hanford	520	26	5.0	494
06/10/01	McNary	520	24	4.6	496
06/11/01	Hanford	493	12	2.4	481
06/11/01	McNary	503	8	1.6	495
06/12/01	Hanford	511	23	4.5	488
06/12/01	McNary	512	28	5.5	484
06/27/01	Hanford	536	16	3.0	520
06/27/01	McNary	554	22	4.0	532
06/28/01	Hanford	532	7	1.3	525
06/28/01	McNary	531	5	0.9	526
06/29/01	Hanford	561	41	7.3	520
06/29/01	McNary	556	39	7.0	517
06/30/01	Hanford	563	45	8.0	518
06/30/01	McNary	560	30	5.4	530
07/01/01	Hanford	533	16	3.0	517
07/01/01	McNary	535	19	3.6	516



Table 2.-Results of Kolmogorov-Smirnov tests for differences in the release-size distributions between PIT-tagged subyearling fall chinook salmon detected and not detected at McNary Dam during 2001. The fork length (FL) with standard deviation shown in parentheses, and the number of fish either detected or not detected at McNary Dam are shown with percentages in parentheses. The maximum difference between the detected and non-detected distributions is shown along with the probability (*P*) of observing this difference if the fish were from identical distributions. A *P* value < 0.05 is considered unlikely.

Release	Mean FL (mm)	Number detected	Number not detected	Maximum difference	<i>P</i>
<b>Hanford Early</b>					
1	66.2 (3.9)	103 (30.0)	240 (70.0)	0.2006	0.0050
2	65.0 (3.5)	126 (24.1)	397 (75.9)	0.1188	0.1250
3	65.0 (3.6)	120 (24.3)	374 (75.7)	0.1351	0.0661
4	65.8 (4.6)	125 (26.2)	352 (73.8)	0.1650	0.0115
5	65.5 (4.3)	132 (27.0)	357 (73.0)	0.1262	0.0856
<b>McNary Early</b>					
1	66.2 (4.1)	99 (24.5)	305 (75.5)	0.1732	0.0196
2	65.8 (4.0)	117 (22.5)	402 (77.5)	0.1821	0.0042
3	65.4 (3.3)	104 (21.0)	392 (79.0)	0.1760	0.0106
4	66.8 (4.1)	121 (24.6)	370 (75.4)	0.1500	0.0294
5	66.3 (4.6)	120 (24.8)	363 (75.2)	0.2050	0.0008
<b>Hanford Late</b>					
6	67.8 (3.7)	118 (22.3)	412 (77.7)	0.1626	0.0137
7	67.0 (4.2)	113 (21.5)	412 (78.5)	0.1336	0.0771
8	72.3 (4.0)	93 (17.8)	429 (82.2)	0.0854	0.6135
9	72.0 (4.8)	83 (16.0)	435 (84.0)	0.1278	0.1908
10	71.8 (4.3)	94 (18.1)	424 (81.9)	0.2707	0.0000
<b>McNary Late</b>					
6	68.3 (3.9)	93 (17.5)	437 (82.5)	0.2376	0.0003
7	67.2 (4.0)	77 (14.7)	448 (85.3)	0.2031	0.0074
8	70.7 (4.0)	82 (15.9)	434 (84.1)	0.1911	0.0110
9	72.1 (4.5)	103 (19.5)	426 (80.5)	0.1747	0.0109
10	72.0 (4.2)	116 (22.4)	402 (77.6)	0.1850	0.0036

released and subsequently detected at McNary Dam in both the early and late McNary Reservoir release groups. All five individual releases within both of the groups had probabilities <0.05. Both Hanford Reach release groups generally had low probabilities, suggesting a possible size effect, but too few probabilities from individual release group were low enough to suggest a size effect for the entire group.

#### *Detection Probabilities and Survival Estimation*

Survival estimates for subyearlings released in 2001 ranged from 0.49 to 0.26 (Table 3). Mean survival was highest for fish released in early June in the Hanford Reach (weighted mean=0.41, SE=0.0198) and lowest for the McNary Reservoir releases in late June (weighted mean=0.32, SE=0.0180). Mean survival of fish released in the Hanford Reach was significantly higher than of fish released in McNary Reservoir both early ( $Z=1.71$ ,  $P=0.0864$ ) and late ( $Z=2.75$ ,  $P=0.0062$ ) in the year (Table 4). A statistically significant seasonal effect within release location was not observed in 2001.

Detection probabilities of individual release groups at McNary Dam ranged from 0.68 for the last McNary Reservoir release in early June to 0.27 for the fourth release in the Hanford Reach in late June (Table 3). The joint survival and detection probabilities ( $\lambda$ ) of individual release groups below McNary Dam in 2001 ranged from 0.31 to 0.18, and averaged 0.29 for the early Hanford groups, 0.26 for the early McNary groups, 0.24 for both of the late Hanford and McNary groups (Table 3). Because of the low number of detections at points downstream of John Day Dam, we were unable to test whether detection at McNary Dam affected subsequent detection at John Day Dam.

Arrival dates at McNary Dam of individual release groups were not homogenous for either the early or late Hanford releases based on Pearson's chi-square tests. However, the individual groups released in McNary Reservoir did exhibit homogeneous arrival dates within their respective time periods (e.g., early and late June). Within time periods, Hanford Reach releases did not exhibit homogeneous mixing with the McNary Reservoir release groups based on the chi-square tests. However, visual inspection of plots of arrival dates at McNary Dam showed that fish released during the same time period but at different locations arrived within a couple of days of each other (Figures 2-3). The early and late releases had distinctly different arrival distributions.

Table 3.-Summary of Hanford Reach subyearling fall chinook salmon survival estimates, detection probabilities, and joint survival and detection probabilities ( $\lambda$ ) calculated for each release in 2001. Standard errors are shown in parentheses. The weighted mean and the standard error are also reported for each group of releases.

Release group	Survival probability to McNary Dam	Detection probability at McNary Dam	Detection and survival ( $\lambda$ ) below McNary Dam
<b>Hanford Early</b>			
1	0.4959 (0.0732)	0.4500 (0.0725)	0.2763 (0.0513)
2	0.3802 (0.0484)	0.4828 (0.0656)	0.2917 (0.0463)
3	0.4011 (0.0549)	0.4492 (0.0665)	0.2841 (0.0480)
4	0.3845 (0.0453)	0.5570 (0.0679)	0.2929 (0.0457)
5	0.4311 (0.0531)	0.4706 (0.0628)	0.3053 (0.0472)
Weighted mean	0.4140 (0.0198)		0.2912 (0.0047)
<b>McNary Early</b>			
1	0.3805 (0.0560)	0.5191 (0.0798)	0.2564 (0.0494)
2	0.3219 (0.0386)	0.5686 (0.0694)	0.3053 (0.0472)
3	0.4194 (0.0796)	0.3846 (0.0779)	0.1875 (0.0436)
4	0.4465 (0.0737)	0.4525 (0.0787)	0.1818 (0.0388)
5	0.3237 (0.0360)	0.6765 (0.0712)	0.2762 (0.0436)
Weighted mean	0.3607 (0.0240)		0.2559 (0.0234)
<b>Hanford Late</b>			
6	0.3827 (0.0559)	0.4573 (0.0711)	0.2500 (0.0462)
7	0.4210 (0.0706)	0.3394 (0.0628)	0.2602 (0.0514)
8	0.4337 (0.1024)	0.2926 (0.0740)	0.1692 (0.0465)
9	0.3514 (0.0772)	0.2747 (0.0669)	0.2500 (0.0625)
10	0.3495 (0.0654)	0.3764 (0.0759)	0.2308 (0.0523)
Weighted mean	0.3872 (0.0157)		0.2386 (0.0141)
<b>McNary Late</b>			
6	0.3020 (0.0511)	0.4357 (0.0784)	0.2537 (0.0532)
7	0.2595 (0.0530)	0.4615 (0.0978)	0.1905 (0.0495)
8	0.2940 (0.0563)	0.4079 (0.0830)	0.2333 (0.0546)
9	0.3499 (0.0572)	0.3991 (0.0707)	0.2602 (0.0514)
10	0.3583 (0.0499)	0.4976 (0.0728)	0.2584 (0.0463)
Weighted mean	0.3217 (0.0180)		0.2449 (0.0115)

Table 4.-Paired comparisons of 2001 survival of subyearling fall chinook salmon release groups using two-tailed Z-tests on the weighted mean and SE of the survival estimates. The probability ( $P$ ) that the two means were drawn from the same normal distribution of means is presented and significance is assumed at  $P < 0.10$ .

Release Groups	Weighted		Z	$P$
	mean	Weighted SE		
Early Hanford	0.4140	0.0198	1.7145	0.0864
Early McNary	0.3607	0.0240		
Late Hanford	0.3872	0.0157	2.7457	0.0062
Late McNary	0.3217	0.0180		
Early Hanford	0.4140	0.0198	1.0603	0.2892
Late Hanford	0.3872	0.0157		
Early McNary	0.3607	0.0240	1.3024	0.1928
Late McNary	0.3217	0.0180		

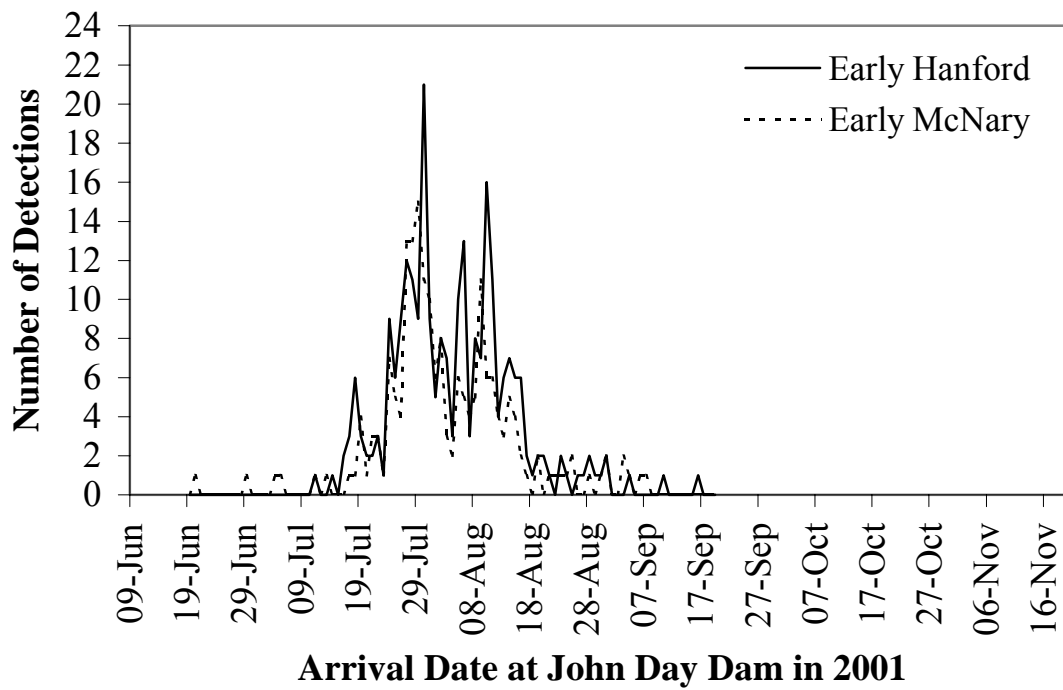
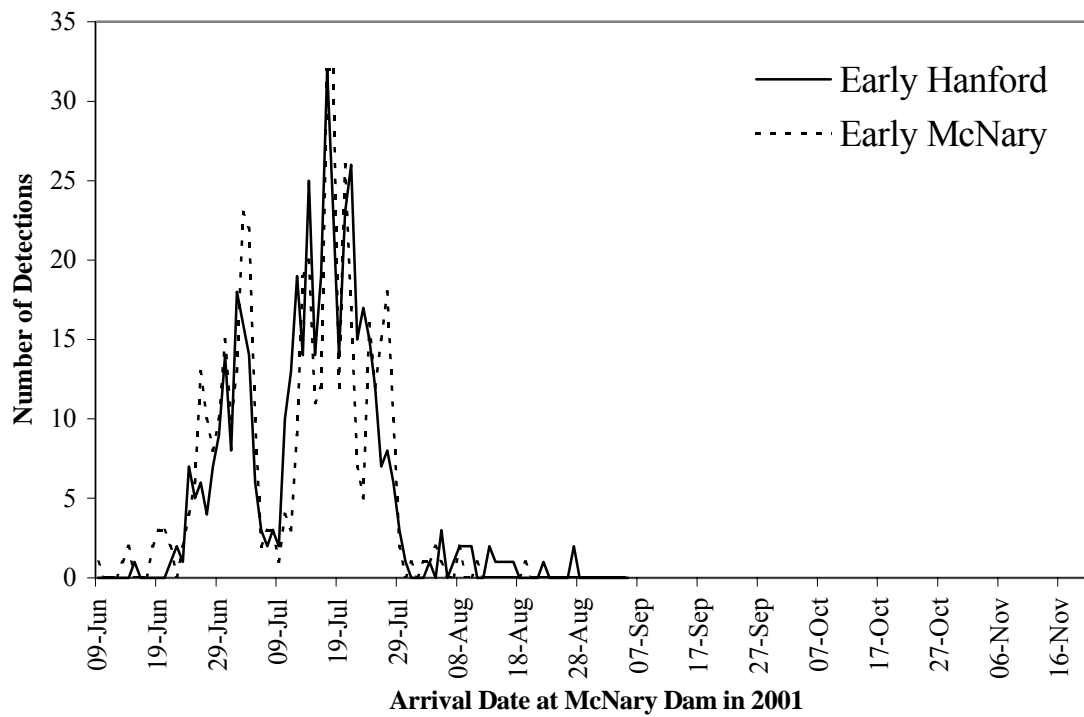


Figure 2.-Plots of the 2001 arrival distributions at McNary and John Day dams from the early Hanford Reach and McNary Reservoir releases of PIT-tagged fall chinook salmon.

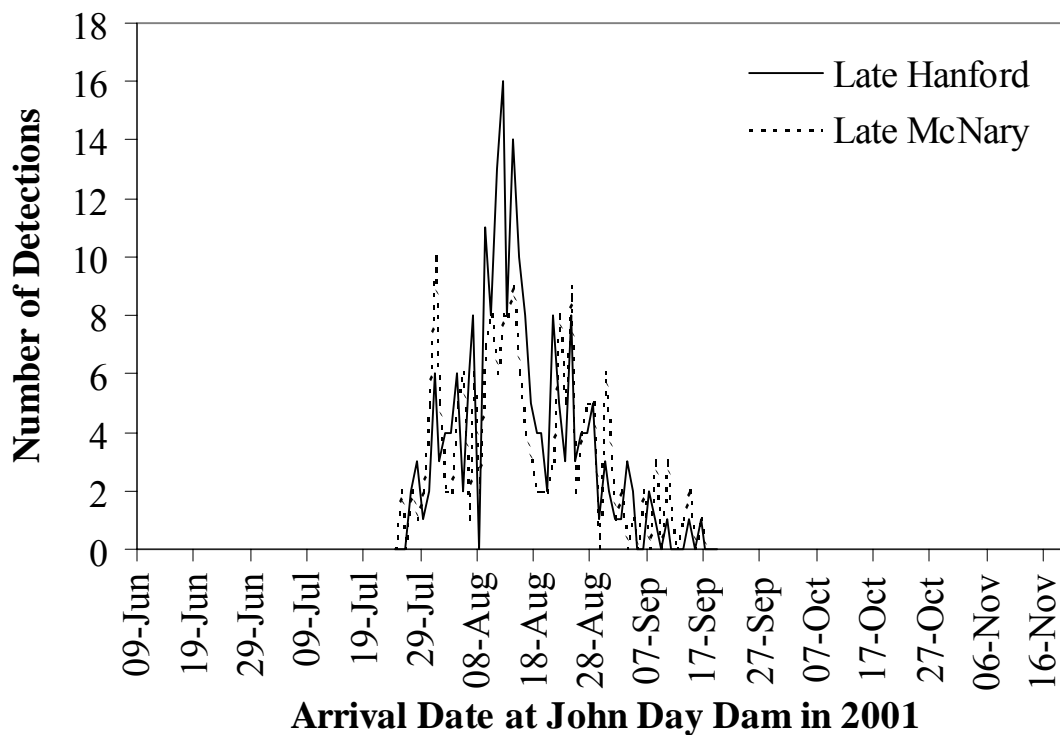
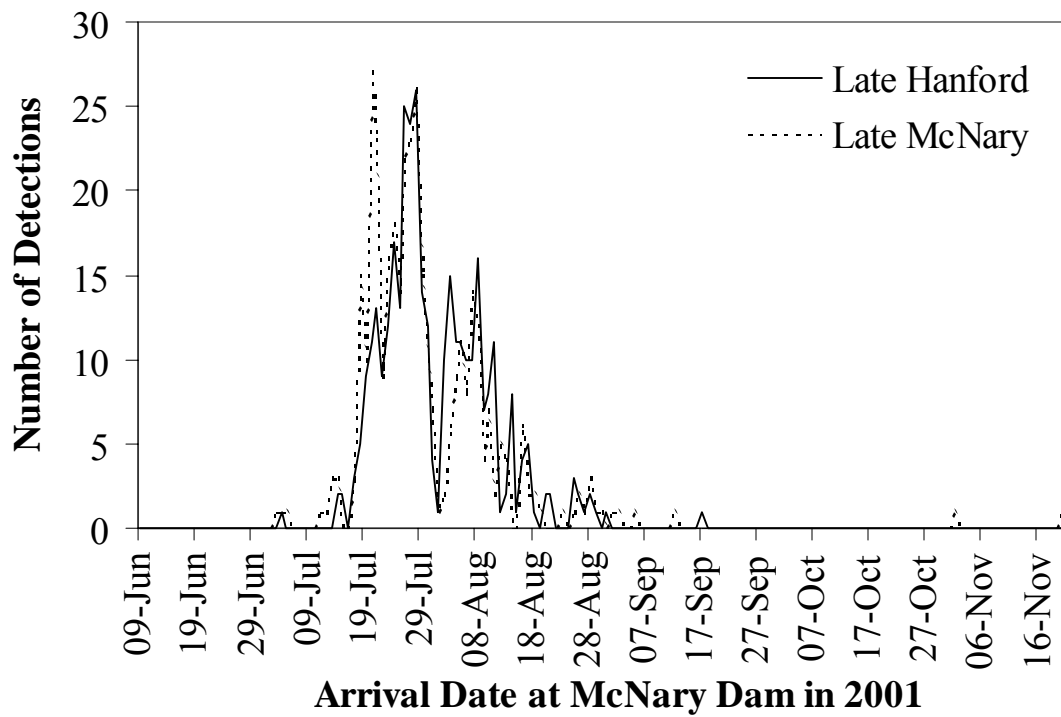


Figure 3.-Plots of the 2001 arrival distributions at McNary and John Day dams from the late Hanford Reach and McNary Reservoir releases of PIT-tagged fall chinook salmon.

Travel time of subyearlings from release sites to McNary Dam decreased throughout the season. The median travel time of subyearlings from the Hanford Reach to McNary Dam was 35.8 d (range=32.6-38.7 d) for fish released in early June, and was 30.1 d (range=27.0-35.2 d) for fish released in late June (Table 5). The median travel time of subyearlings from the head of McNary Reservoir to McNary Dam was 34.9 d (range=31.1-35.1 d) for fish released in early June, and was 27.9 d (range=26.0-29.3 d) for fish released in late June (Table 5). Priest Rapids Dam discharge averaged 81 and 63 kcfs for the median time fish were at large for the early and late releases, respectively (Figure 4). Fish released earlier in the season generally took longer to travel to McNary Dam than fish released later. It only took about 1-3 d longer for subyearlings to travel to McNary Dam from the Hanford Reach than from the head of McNary Reservoir in spite of the greater distance that the Hanford Reach added. PIT-tagged fish released in 2000 also had slower travel times for Hanford-released fish compared to fish released in McNary Reservoir, but fish generally traveled at similar rates regardless of when they were released (Table 6, unpublished data). Travel times were slightly faster in 2000 compared to 2001. In 2001, there were only weak negative correlations between subyearling travel times and time of release (range=0.25-0.46; Figure 5).

## Discussion

Our tagging mortalities were higher than those reported in other studies dealing with juvenile salmon. Our mortality averaged 3.6% compared to 1%, or less, reported for PIT tagging both hatchery and wild fish in the Snake River (Muir et al. 1998; W. Connor, personal communication). However, our mortalities were not much different than the direct mortality of subyearlings fall chinook salmon PIT tagged by WDFW in the Hanford Reach, which was 2.3% in 1995, 5.4% in 1996, and 4.3% in 1997 (Wagner 1996; Hillson et al. 1997 and 1998). Our mortality numbers are the sum of mortalities from all sources such as handling, tagging, and 24-h delayed mortality. Most fish PIT-tagged in field operations are released within an hour or less after tagging so mortality rates represent only those from tagging. Our higher number of mortalities may have been due to the condition of fish at time of tagging. Although we culled out every fish with any suspected problems, some fish in the population did appear to have tail fungus and this may have contributed to delayed mortality. However, visual inspection of plots of the short-term tagging mortality of individual release

Table 5.-Travel time (d) statistics for the groups of PIT-tagged subyearling fall chinook salmon released in the Hanford Reach and McNary Reservoir and detected at McNary Dam in 2001.

Release group	Date	N	Min.	20%	Med.	80%	Max.	Mean	SD
<b>Hanford Early</b>									
1	6/07/01	75	15.9	24.0	32.6	40.5	61.1	32.6	10.3
2	6/09/01	96	5.8	31.8	38.7	44.5	68.3	37.9	10.9
3	6/10/01	89	12.1	23.5	37.1	41.9	67.9	35.2	10.7
4	6/11/01	104	10.2	28.7	35.3	41.6	77.2	35.3	11.0
5	6/12/01	99	12.1	20.0	34.1	39.8	76.3	32.0	10.8
Combined		463	5.8	23.7	35.8	41.8	77.2	34.7	11.0
<b>McNary Early</b>									
1	6/07/01	80	1.6	21.4	31.1	43.6	61.6	32.6	12.8
2	6/09/01	95	4.8	20.9	35.1	42.2	71.2	33.5	12.4
3	6/10/01	80	7.7	23.5	34.8	41.8	55.8	34.8	9.9
4	6/11/01	100	9.5	22.1	33.5	39.9	52.6	31.5	9.8
5	6/12/01	106	6.2	18.9	34.7	41.5	47.4	31.4	10.8
Combined		461	1.6	21.8	34.9	41.8	71.2	32.7	11.2
<b>Hanford Late</b>									
6	6/27/01	90	7.2	25.3	30.2	39.8	54.7	32.1	8.8
7	6/28/01	75	16.1	25.8	32.3	40.8	61.3	34.4	10.0
8	6/29/01	65	19.0	22.5	29.5	38.5	63.0	32.0	8.9
9	6/30/01	50	18.0	27.1	35.2	42.4	56.2	35.0	9.4
10	7/01/01	68	17.9	23.0	27.0	37.6	78.3	31.0	10.8
Combined		348	7.2	25.2	30.1	40.4	78.3	32.8	9.6
<b>McNary Late</b>									
6	6/27/01	70	6.0	23.3	29.3	38.9	146.2	32.3	17.3
7	6/28/01	63	7.5	12.6	29.0	50.4	75.9	31.4	11.5
8	6/29/01	62	19.0	22.0	29.0	49.6	19.0	33.8	15.4
9	6/30/01	74	13.9	20.1	27.8	49.9	63.9	29.9	11.2
10	7/01/01	92	12.0	19.8	26.0	34.7	66.2	27.5	8.9
Combined		361	6.0	21.5	27.9	38.8	146.2	30.7	13.1



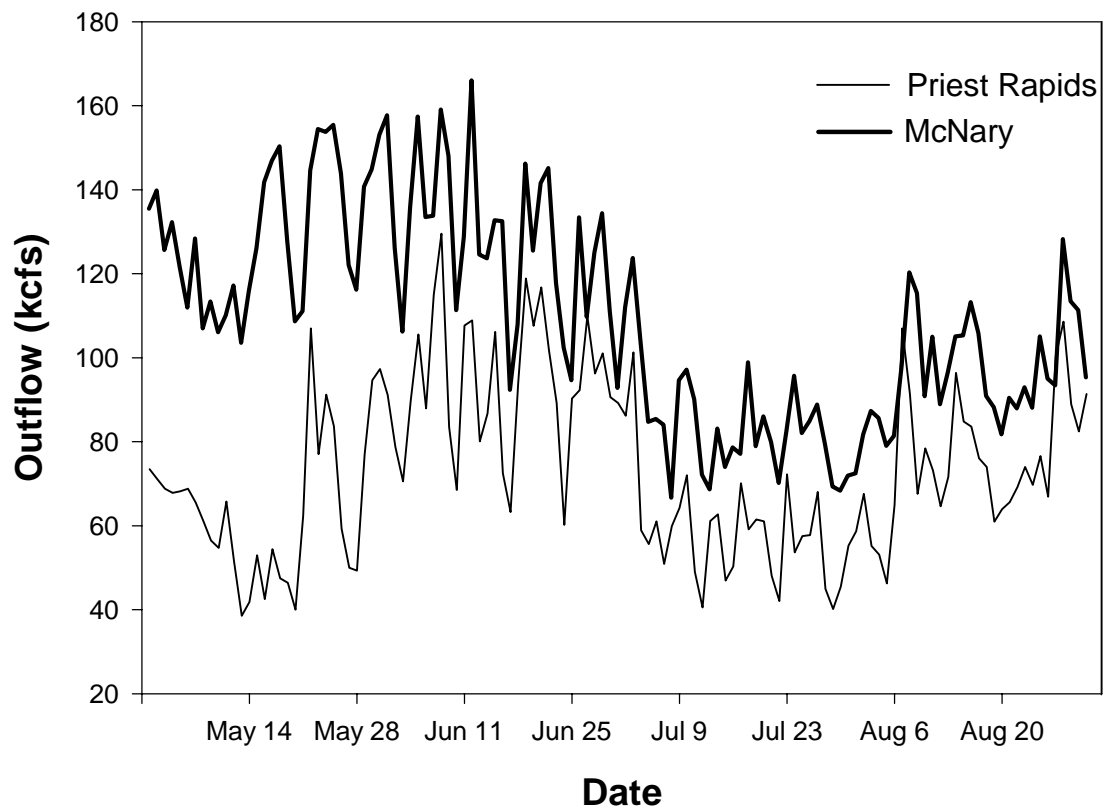


Figure 4.-Outflow from Priest Rapids Dam and McNary Dam during the Hanford Reach survival study 2001.

Table 6.- Travel time (d) statistics for the groups of PIT-tagged subyearling fall chinook salmon released in the Hanford Reach and McNary Reservoir and detected at McNary Dam in 2000.

Release group	Date	N	Min.	20%	Med.	80%	Max.	Mean	SD
<b>Hanford Early</b>									
1	6/01/00	72	20.3	25.7	30.9	36.8	50.1	31.9	7.1
2	6/02/00	128	7.3	23.9	30.6	35.0	71.8	30.1	9.4
3	6/03/00	143	17.7	25.9	32.0	36.5	74.4	32.1	7.6
4	6/04/00	147	18.1	26.1	31.3	35.9	61.0	32.1	7.8
5	6/05/00	133	11.8	26.4	30.8	36.3	60.7	31.5	8.2
Combined		623	7.3	25.7	31.2	36.1	74.4	31.6	8.1
<b>McNary Early</b>									
6	6/09/00	153	14.0	25.1	29.2	34.2	64.9	31.2	8.9
7	6/10/00	143	8.6	23.2	28.9	41.8	70.8	31.6	11.0
8	6/11/00	108	17.5	20.7	25.4	29.4	57.9	31.7	8.6
9	6/12/00	144	9.3	22.1	27.8	38.7	52.8	29.5	9.9
10	6/13/00	141	8.5	21.7	27.7	38.9	58.0	29.6	10.3
Combined		689	8.5	23.4	28.5	39.5	70.8	30.7	9.8
<b>Hanford Middle</b>									
11	6/17/00	139	10.4	23.8	33.4	40.8	59.3	32.3	10.1
12	6/18/00	143	6.1	22.7	33.3	40.8	58.1	32.3	11.1
13	6/19/00	169	9.7	20.7	31.7	41.5	60.1	31.4	10.8
14	6/20/00	122	13.0	21.0	33.8	41.3	55.1	31.3	9.9
15	6/21/00	119	12.8	21.0	33.6	40.1	50.2	31.9	8.9
Combined		692	6.1	21.4	32.8	40.6	60.1	31.8	10.2
<b>Hanford Late</b>									
16	6/29/00	40	5.2	15.1	25.4	30.0	41.6	24.0	8.8
17	6/30/00	101	5.2	11.8	30.5	34.9	80.5	27.5	12.9
18	7/01/00	100	7.2	27.8	33.6	39.9	51.9	33.1	9.7
19	7/02/00	94	8.1	24.8	31.6	38.5	68.7	31.7	9.7
20	7/03/00	63	6.0	11.0	25.7	31.9	46.7	24.1	11.1
Combined		398	5.2	20.0	30.9	38.0	80.5	29.0	11.3

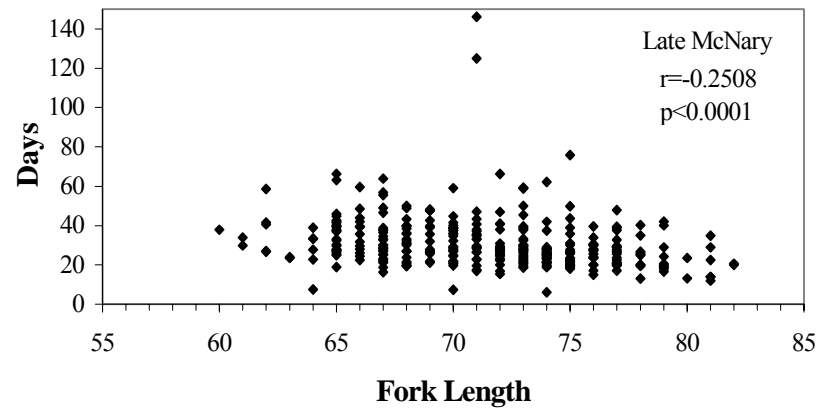
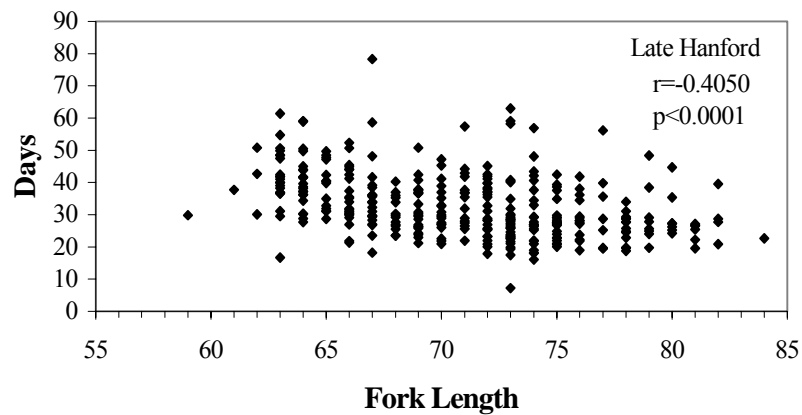
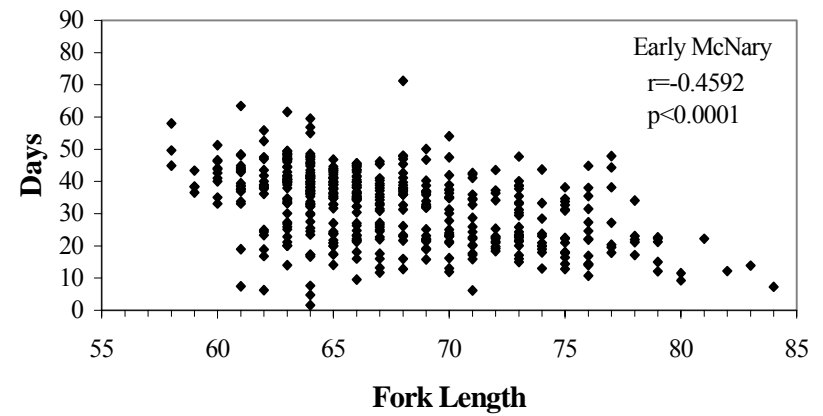
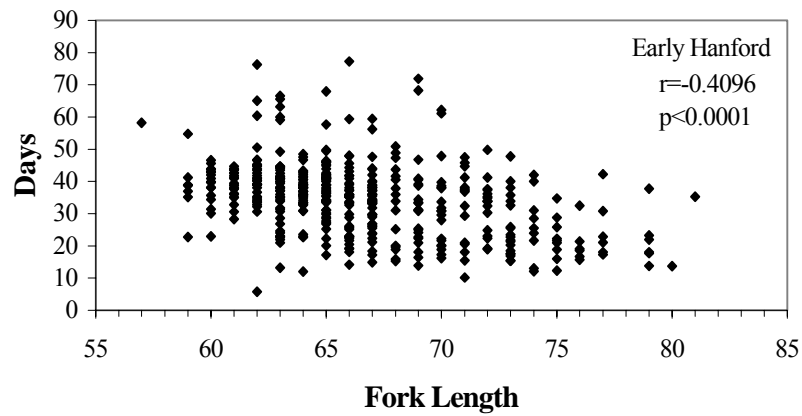


Figure 5.- Bivariate plots of the number of days to detection at McNary Dam and fork length at release for each group of PIT-tagged subyearling fall chinook salmon in 2001. Results from the bivariate correlation analysis are in the upper right hand corner.

groups and their estimated survival to McNary Dam showed no discernable relationships. As such, we do not believe that our mortality rate in 2001 influenced our survival estimates.

The environment in which subyearlings rear and migrate may affect their survival. We found only small differences in survival between fish released in the Hanford Reach and in McNary Reservoir within each seasonal time period. In fact, fish released further up river in the Hanford Reach had slightly higher survival than fish released in McNary Reservoir in spite of having an additional 70 km to travel before reaching the reservoir. Although Hanford-released fish took 1-3 d longer to reach McNary Dam, this added time did not result in reduced survival compared to the reservoir-released fish. There are at least two possible explanations for these observations. First, fish released in both riverine and reservoir environments may be exposed to the same mortality pressures (e.g., predation), which become lessened as fish grow to migratory size. In this case, survival rates would be similar and the longer travel time of Hanford-released fish would be due to the longer distance to travel to McNary Reservoir. An alternative explanation may be that there were greater mortality pressures in reservoir habitats that affect fish regardless of when they arrive in the reservoir. If Hanford Reach fish were larger when they reach McNary Reservoir because of extended rearing time in riverine habitats, then they may have been able to avoid some of these mortality pressures (e.g., larger fish can better escape predators) resulting in higher survival. If this is true, then it may explain why fish released in the Hanford Reach survived in higher numbers to McNary Dam.

The difference in survival of groups released in the Hanford Reach and McNary reservoir may also result from differences in size dependent mortality in shoreline habitats. In 2001, smaller fish at release in McNary Reservoir had lower detection frequency relative to the size distribution of the entire release, but this was not observed for the Hanford Reach releases. Because the paired releases were comprised of similar-sized fish with nearly the same arrival times at McNary Dam, we do not believe that the Hanford Reach-released fish were different enough in size to have contributed to detection differences. Thus, we believe that the detection differences were due to differential survival between groups released at the two locations. In addition, relatively small individuals released in McNary Reservoir may not have experienced the same survival as small individuals released in the Hanford Reach.

Survival was higher for fish released early in the season compared to fish released later within release locations. This seasonal effect on survival has also been observed for both hatchery and natural subyearling fall chinook salmon tagged and released in the Snake River (Connor et al. 1998a; Smith et al. 2002; Connor et al. Chapter One, this report). It is often suggested that earlier migrating fish have higher survival because river flows are higher and temperatures are cooler than they are later in the season. Indeed this was true in 2001, a drought year, when Priest Rapids Dam flows averaged 81 and 63 kcfs for the median time fish were at large for the early and late releases, respectively. Flows in 2001 were the second lowest on record. However, late-released fish traveled to McNary Dam faster than early-released fish under lower flows, but this may be the result of the inverse relationship between fish size and travel time (Connor et al. Chapter One, this report). Fall chinook salmon have a complex life history in regards to their rearing and migratory behavior. It is unknown whether fish rear to a specific size and then exhibit directed downstream migration or whether they continually move downstream as they rear. This complicates establishing relationships between juvenile fall chinook salmon survival and flow and temperature because fish are exposed to a range of conditions between their release and subsequent detection. However, higher water velocities in the Hanford Reach may have contributed to the higher survival of fish released there by reducing their exposure time to predators.

Our survival estimates for Columbia River fall chinook salmon compare well with those obtained for cohorts of naturally-produced Snake River fall chinook salmon in 2001 (Connor et al. Chapter 1 in this report). The survival of Snake River fall chinook salmon from the Hells Canyon Reach to Lower Granite Dam ranged from 0.410 for the earliest emerging cohort to 0.04 for the latest cohort and averaged 0.205 overall. Our estimates ranged from 0.414 to 0.322 (mean=0.371), which are comparable to the high end of the Snake River estimates. Survival in 2001 in the Snake River was the lowest observed for the years 1998-2001, and was likely due to the very low flows and high temperatures that were prevalent during that drought year. Survival of fall chinook salmon in the Snake River in more typical years (1998-2000) ranged from a high of 0.877 to a low of 0.356, and average survival ranged from 0.477 to 0.706. The drought conditions in 2001 likely contributed to the relatively low survival in the Columbia River as well.

The results of this study suggest that the success of the Hanford stock is not a result of high survival prior to encountering the first dam. If survival is 30-40% for 62-85 mm fish (the size we tagged), then one might expect even lower survival for fish 38 mm (emergence size) to 62 mm. The question arises as to why the survival of juvenile fall chinook salmon was so low in 2001? Our estimates of 0.30-0.40 were substantially lower than the estimated survival for fish traveling from McNary Dam to John Day Dam in 2001, which was 0.581 (Smith et al. 2002). Power peaking operations at Priest Rapids Dam often result in high mortality of juvenile fish in the Hanford Reach due to stranding and entrapment. The estimated mortality due to stranding in 2001 was about 1.6 million fish—the highest since estimates began to be generated in 1999 (Nugent et al. 2002). Adding to low survival in 2001 were low river flows, the second lowest on record. As previously mentioned, drought conditions are characterized by low flows, high water temperatures, and slow travel times, which have been linked to low survival (Connor et al. Chapter 1 in this report). The cumulative losses from passage through reservoirs downstream of McNary Dam result in an even lower proportion of the annual Hanford Reach production to survive to the estuary. However, it is apparent that enough fish survive to return as adults to maintain the health of the Hanford Reach stock.

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## Journal Manuscripts

### **CHAPTER THREE**

#### Estimating the Carrying Capacity of the Snake River for Fall Chinook Salmon Redds\*

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*Abstract.*-Recovery planning for imperiled populations of anadromous salmonids can require estimates of the carrying capacity of a river for redds (hereafter, redd capacity). We estimated redd capacity for the 106 known fall chinook salmon spawning sites in the upper and lower reaches of the Snake River. We used a modification of the Instream Flow Incremental Methodology to estimate spawning area ( $m^2$ ) for 12 representative study sites. We estimated that one redd occupied  $70 m^2$  of spawning area at the most heavily utilized site. Spawning area was estimated at the 12 study sites using a stable flow that was implemented to prevent redd de-watering, and two other flows that encompassed natural fluctuation. We estimated redd capacity for each study site by dividing the amount of spawning area modeled at each of the three flows by  $70 m^2$ . We input the redd capacity estimates for the study sites into the equation for a stratified random sample to make three estimates of redd capacity for all 106 known spawning sites. The estimates ranged from 2,446 to 2,570 redds. We conclude that the Snake River can support the 1,250 redds needed to satisfy Endangered Species Act de-listing criteria. However, annual surveys should be conducted to eventually determine if recruitment efficiency is affected by density dependent factors before the recovery goal is achieved.

## Introduction

The construction of hydroelectric and diversion dams has eliminated or reduced spawning habitat used by anadromous salmonids in the Pacific Northwest (Wunderlich et al. 1994, Kondolf et al. 1996, Dauble and Geist 2000). Spawning habitat loss is one factor for the imperiled status of many anadromous salmonid stocks. The development of recovery plans for imperiled stocks sometimes requires estimating the number of redds that existing or lost habitat can carry (hereafter, redd capacity). This was the case with Snake River fall chinook salmon (*Oncorhynchus tshawytscha*), a stock that was listed as threatened under the Endangered Species Act (ESA) in 1992 (National Marine Fisheries Service 1992).

S Snake River fall chinook salmon were displaced from the historic spawning area near Marsing, Idaho (Groves and Chandler 1999) by the construction of Brownlee, Oxbow, and Hells Canyon dams (Figure 1). By 1975, Lower Granite, Little Goose, Lower Monumental, and Ice Harbor dams impounded the lower 224 km of the Snake River leaving approximately 173 km of riverine spawning habitat between Hells Canyon Dam and the upper end of Lower Granite Reservoir (hereafter, the Snake River)(Figure 1). An estimate of redd capacity was needed to help define a recovery goal to match the remaining habitat. Few empirical data were available when the recovery plan was drafted, however, and biologists relied heavily on professional judgement to establish a proposed recovery goal of 2,500 adults (National Marine Fisheries Service 1995). The recovery goal equates to a redd capacity of 1,250 assuming an equal sex ratio for spawners.

While the Snake River fall chinook salmon recovery plan was being developed, we began to study spawners and their habitat. Water flow from Hells Canyon Dam was also stabilized at approximately 260 m<sup>3</sup>/s during the spawning and incubation seasons to prevent redd de-watering (Groves and Chandler 1999). In this chapter, we use data that were collected after the proposed recovery plan for Snake River fall chinook salmon was written to estimate fall chinook salmon redd capacity under a stable flow regime for two reaches of the Snake River. We also discuss the suitability of the proposed recovery goal (1,250 redds) in light of the redd capacity estimates we generate.

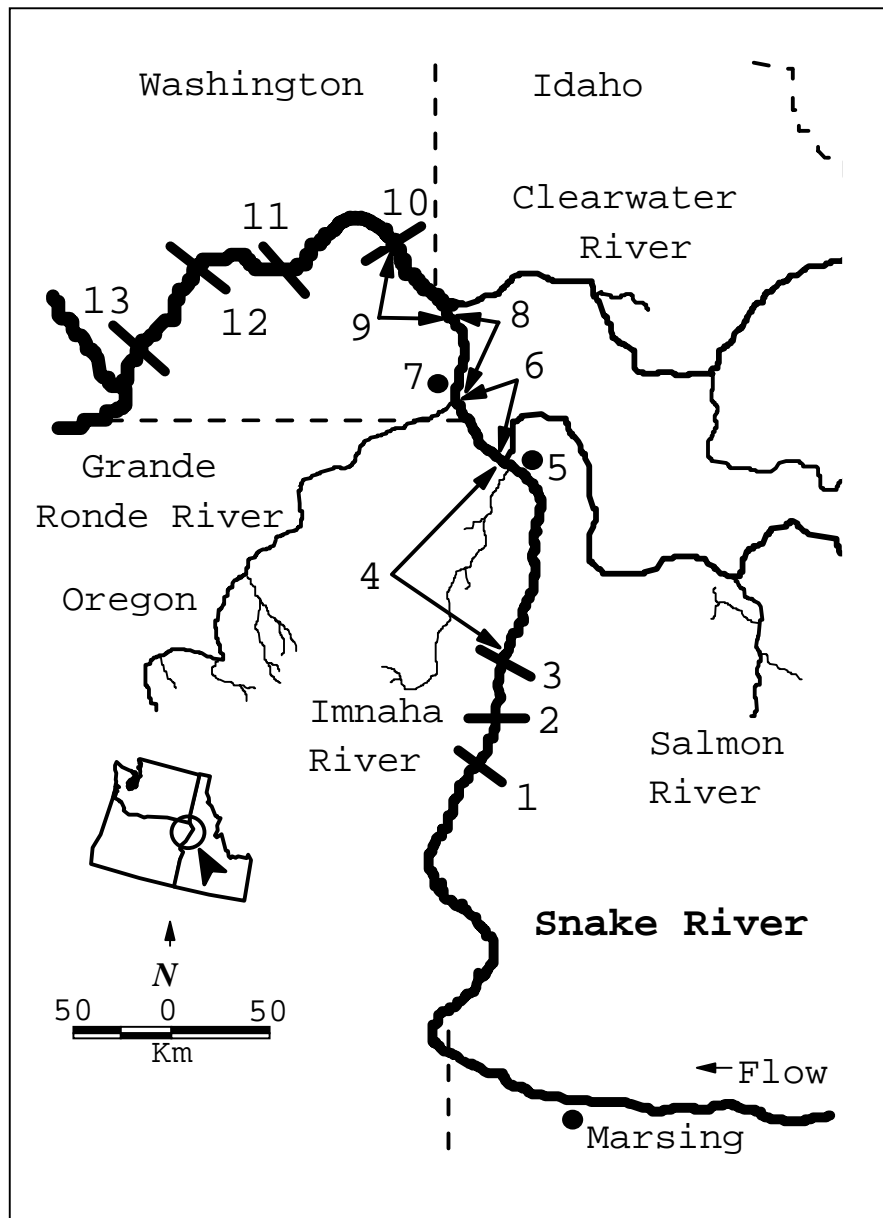


Figure 1.-The Snake River including the locations of the upper, middle, and lower reaches, and the historic spawning area near Marsing (approximately rkm 685), major tributaries, dams and U.S. Geological Survey gaging stations. The locations referenced by number are 1) Brownlee Dam; 2) Oxbow Dam; 3) Hells Canyon Dam; 4) Upper Reach Snake River; 5) Site 311.5; 6) Middle Reach Snake River; 7) Anatone, Washington; 8) Lower Reach Snake River; 9) Lower Granite Reservoir; 10) Lower Granite Dam; 11) Little Goose Dam; 12) Lower Monumental Dam; and 13) Ice Harbor Dam.

## Study Area

For a detailed description of the Snake River, we refer the reader to papers by Groves and Chandler (1999) and Dauble and Geist (2000). The Snake River can be divided into three reaches (Figure 1) based on differences in channel morphology and discharge. The volume of water flowing through the upper reach is controlled by releases of water from the Hells Canyon Dam (Groves and Chandler 1999). The Imnaha, Salmon, and Grande Ronde Rivers (Figure 1) provide additional water to the lower reach of the Snake River and cause natural flow fluctuation during spawning.

Between 1986 and 2000, there were 78 documented spawning sites in the upper reach, 11 in the middle reach, and 28 in the lower reach (Garcia et al. 2001). Spawning sites were defined as areas where redds occurred within a relatively contiguous patch of medium gravel to small cobble (long-axis diameter 2.6 to 15.0 cm; Groves and Chandler 1999).

## Methods

*Study sites and maps.*—From 1991 to 1994, we selected five known spawning sites for study in the upper reach and seven in the lower reach. We did not select any study sites in the middle reach because of low spawner use from 1991 to 1994. We established one primary transect (Figure 2) at 11 of our study sites to represent the habitat used by fall chinook salmon spawners. Three primary transects were established at the largest and most complex study site at river km (rkm) 266.5. The locations of the 14 primary transects were surveyed using an electronic total station.

We also established numerous supplemental transects at approximately 15-m intervals upstream and downstream of primary transects to bound the spawning habitat. We used an 8 mm video camera positioned 1.2 m above the ground to record substrate above the water line along each primary and supplemental transect. Mean long-axis diameter of the dominant substrate was assessed visually in water less than 0.6-m deep. We used an underwater video camera to tape substrate images in water > 0.6-m deep (Groves and Garcia 1998). At least 20 substrate measurements were made per transect (Geist et al. 2000) and the measurement locations and channel elevations were surveyed using the total station. We determined the mean long-axis diameter of the dominant substrate in each video image (Groves and Chandler 1999).

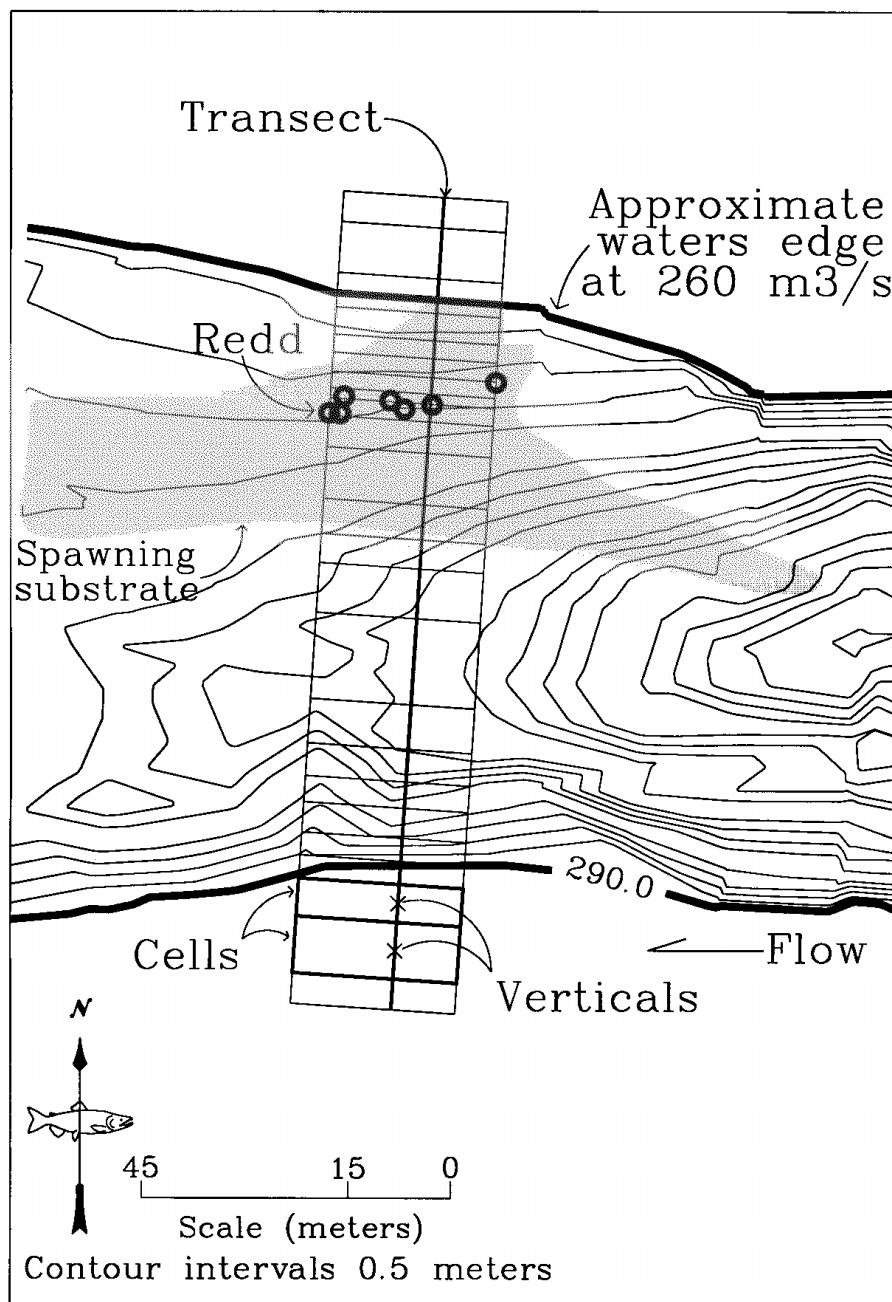


Figure 2.— The study site at rkm 311.5 including the location of the spawning substrate patch, primary transect, verticals, cell boundaries, and fall chinook salmon redds.



We made bathymetric maps of each study site (e.g., Figure 2) by inputting the substrate measurement and channel elevation locations into AutoCAD® and Softdesk® mapping software. These maps included the distribution of substrate with long-axis diameters ranging from 2.6 to 15.0 cm (hereafter, spawning substrate patches) and the locations of redds we surveyed between 1991 and 1994.

*Estimating spawning area.*—We collected velocity calibration data (Bovee and Milhous 1978) at verticals (Figure 2) spaced along the primary transects using U. S. Geological Survey (USGS) gear or an acoustic Doppler current profiler. We surveyed the location of the verticals using the total station so that verticals could be positioned on the bathymetric maps (Figure 2). Velocity calibration data were usually collected during spawning (flow ranges upper reach = 250 to 300 m<sup>3</sup>/s; lower reach = 290 to 430 m<sup>3</sup>/s). We also collected stage-discharge data (Bovee and Milhous 1978) over a wide range of flows (upper reach 260 to 1,190 m<sup>3</sup>/s; lower reach 280 to 1,300 m<sup>3</sup>/s). All velocity calibration data were collected during periods of stable flow.

We calibrated the hydraulic model IFG-4 (Milhous et al. 1984) to allow the simulation of mean water column velocity at the verticals over the spawning substrate patches at each study site. Velocity adjustment factors were calculated by dividing the simulated flow by the calculated flow to assess model fit. All of the velocity adjustment factors fell in the range of 0.8 to 1.2 indicating IFG-4 fit the data (Bovee and Bartholow 1995). We used stage-discharge regressions developed for the IFG-4 data decks (Milhous et al. 1984) to simulate water depth at the verticals over the spawning substrate patches at each study site. Depth was simulated by subtracting the surveyed channel elevation at each vertical from the predicted water surface elevation.

*Estimating spawning-area-per-redd.*—The IFG-4 model typically represents the stream bed in the form of rectangles called "cells" (Milhouse et al. 1984). We used vertical spacing to determine the width of each cell (Figure 2). We determined cell length two ways. For 11 of the primary transects we based cell length on the maximum distance redds were located up and downstream of the transect (Figure 2). We surveyed one or two redds at the primary transects representing three of our upstream sites, although the substrate patches at these sites were obviously large enough to support additional spawning. We determined cell length at these three sites by using the bathymetric maps to determine the up and downstream distances

within the substrate patch that was represented by the primary transect. We then calculated the area of the spawning substrate in each cell by using AutoCAD® and Softdesk® mapping software.

We used IFG-4 and the stage-discharge regressions to simulate water depth and mean column velocity within each cell under the flow that occurred the year we surveyed the maximum number of redds at each study site. We calculated the simulation flow for this analysis as the average of the daily mean flows between the onset and end of spawning. Daily mean flow records for all simulations were obtained from USGS gages at Hells Canyon Dam (rkm 398.6) in the upper reach and Anatone, Washington (rkm 269.7) in the lower reach (Figure 1).

The cells with spawning substrate were considered to be suitable for spawning if the simulated depths ranged from 0.2 to 6.5 m, and the simulated mean water column velocities ranged from 0.4 to 2.1 m/s (Groves and Chandler 1999). We calculated spawning area ( $\text{m}^2$ ) for each study site by summing the area of spawning substrate in the cells that met the above suitability criteria for water depth and water velocity. We then estimated spawning-area-per-redd at each study site by dividing spawning area by the maximum observed redd count.

*Estimating redd capacity.*—We simulated water depth and mean water column velocity for sites in the upper reach of the Snake River under the stable flow regime (i.e.,  $260 \text{ m}^3/\text{s}$ ). To account for flow fluctuation caused by tributary inflow in the lower reach, we simulated water depth and mean water column velocity at flows of 280, 400, and  $520 \text{ m}^3/\text{s}$ . This range included the minimum and maximum daily mean flows observed in the lower reach during our study. We estimated redd capacity for each site at each simulation flow by dividing spawning area by the minimum value of spawning-area-per-redd calculated as described in the previous section of methods. Finally, we estimated redd capacity with a 95% confidence interval for all 106 known spawning sites in the upper and lower reaches by inputting the redd capacity estimates of the 12 study sites into the equation for a stratified random sample (Krebs 1999).

## Results

Spawning area estimates for the 12 study Snake River sites ranged from 601 to  $13,239 \text{ m}^2$  the year the maximum number of redds was surveyed at each study site (Table 1). Spawning-area-per-redd ranged from 70 to  $683 \text{ m}^2$  (Table 1). We selected  $70 \text{ m}^2$  as the area required by spawners to construct a redd.

Table 1.—Estimates of spawning area (SA) per redd (SA/redd) for 12 fall chinook salmon spawning sites along the upper and lower reaches of the Snake River based on the flow (m<sup>3</sup>/s) during spawning the year the maximum number of redds were counted at each site.

Site (rkm)	Year	Simulation flow (m <sup>3</sup> /s)	SA (m <sup>2</sup> )	Maximum redd count	SA per redd (m <sup>2</sup> )
<b>Upper reach</b>					
311.5	1992	261	662	5	132
311.7	1993	270	601	1	601
312.3	1994	262	1,234	5	247
349.6	1993	270	1,366	2	683
352.8	1994	262	665	2	333
<b>Lower reach</b>					
245.2	1992	380	3,077	7	440
259.0	1993	411	773	11	70
261.3	1991	465	4,977	20	249
266.5	1993	411	13,239	30	441
267.0	1993	411	1,735	4	434
267.8	1993	411	1,412	6	235
267.9	1993	411	1,262	14	90

Spawning area estimated for the study sites under the stable flow regime ranged from 601 to 1,234 m<sup>2</sup> in the upper reach, and from 773 to 13,239 m<sup>2</sup> in the lower reach (Table 2). Redd capacity ranged from 9 to 20 for study sites in the upper reach, and from 11 to 189 for study sites in the lower reach (Table 2). Estimated spawning area and redd capacity increased for the lower reach sites at rkm 245.2 (up 21 redds), rkm 266.5 (up 2 redds), rkm 267.8 (up 5 redds), and rkm 267.9 (up 3 redds) as the simulation flow increased from 280 to 520 m<sup>3</sup>/s (Table 2).

The information required for estimating the redd capacity of the 106 known spawning sites in the upper (n = 78) and lower (n = 28) reaches of the Snake River is given in Table 2. The three estimates of redd capacity were 2,446+1,439 (upper reach flow = 260 m<sup>3</sup>/s; lower reach flow = 280 m<sup>3</sup>/s), 2,558+1,427 (upper reach flow = 260 m<sup>3</sup>/s; lower reach flow = 400 m<sup>3</sup>/s), and 2,570+1,421 (upper reach flow = 260 m<sup>3</sup>/s; lower reach flow = 520 m<sup>3</sup>/s).

## **Discussion**

### *Assumptions and Limitations*

We assumed that redd capacity increases as spawning area increases. A correlation analysis between spawning area and maximum redd count would test this assumption. Gallagher and Gard (1999) reported a significant correlation between chinook salmon spawner density and an estimate of spawning area called weighted usable area (Bovee 1982). We did not conduct a correlation analysis because spawner number was critically low, thus the majority of the study sites was under utilized. Fall chinook salmon redds counted during aerial surveys increased from 41 in 1991 to 255 in 2000 (Garcia et al. 2001). We may have an opportunity to validate our redd capacity estimates if adult fall chinook salmon escapement to the Snake River continues to increase.

We equated the recovery goal of 2,500 adults to the Snake River spawning grounds to a redd capacity of 1,250 assuming an equal sex ratio. The information on the sex ratio of wild Snake River fall chinook salmon spawners was inadequate for our modeling because it is limited to small samples of carcasses collected haphazardly during spawning surveys. However, the Washington Department of Fish and Wildlife propagates hatchery Snake River fall chinook that are phenotypically and genetically similar to wild fish (Bugert et al. 1995, Marshall et al. 2000).

Table 2.-Estimates of redd capacity for 12 fall chinook salmon spawning sites along the upper and lower reaches of the Snake River based a stable flow of 260 m<sup>3</sup>/s in the upper reach, and a range of flows in the lower reach of 280, 400, and 520 m<sup>3</sup>/s. The statistics for estimating total redd capacity for the 106 known spawning sites in the upper (n = 78) and lower reaches (n = 28) are also given.

Site (rkm)	Spawning area (m <sup>2</sup> ) by flow (m <sup>3</sup> /s)				Redd capacity by flow (m <sup>3</sup> /s)			
	260	280	400	520	260	280	400	520
<b>Upper reach</b>								
311.5	662	-	-	-	9	-	-	-
311.7	601	-	-	-	8	-	-	-
312.3	1,234	-	-	-	18	-	-	-
349.6	1,142	-	-	-	20	-	-	-
352.8	664	-	-	-	10	-	-	-
n					5			
Sample mean					13.2			
Sample variance					28.7			
<b>Lower reach</b>								
245.2	-	1,876	3,387	3,387	-	27	48	48
259.0	-	773	773	773	-	11	11	11
261.3	-	4,977	4,977	4,977	-	71	71	71
266.5	-	13,105	13,239	13,239	-	187	189	189
267.0	-	1,735	1,735	1,735	-	25	25	25
267.8	-	1,067	1,412	1,412	-	15	20	20
267.9	-	1,262	1,262	1,475	-	18	18	21
n					7	7	7	7
Sample mean					50.6	54.7	55.0	
Sample variance					4,022.0	3,948.3	5,913.0	

The sex ratio observed for spawners at this hatchery between 1988 and 1996 averaged 0.7 females to 1.0 males (Mendel et al. 1992, 1996). We used a 1.0 to 1.0 ratio to simplify our analysis, and to add a measure of conservatism to our redd capacity estimates.

We expanded the measurements taken at 12 spawning sites to all 106 spawning sites, thereby assuming that redd capacity of study sites represented redd capacity of non-study sites. We sampled approximately 10% of the known spawning sites, which we believe represented the spawning habitat at non-study sites. However, redd capacity within study sites was variable as shown by the relatively wide 95% confidence intervals on our redd capacity estimates. We recommend studying additional sites if future research opportunities become available.

We did not measure factors affecting redd capacity such as inter-gravel flow (Burner 1951, Geist and Dauble 1998, Geist 2000), substrate movement, or substrate recruitment. We assumed that inter-gravel flow would not limit redd capacity or cause variability in redd capacity between sites with the same amount of modeled spawning area. We also assumed that substrate movement and recruitment were in dynamic equilibrium. These are strong assumptions that should be tested in the future at both the spatial and temporal scales.

We did not report redd capacity estimates for extreme flow conditions because data were not available to fit the stage-discharge regression required to run IFG-4. Within the range of flows modeled, we found that redd capacity decreased moderately in the lower reach of the Snake River as flow decreased. This suggests that the amount of spawning area might limit redd construction at some low flow level, which in turn could have a temporal effect on production by reducing the number of returning spawners 4 to 5 years later. Stage-discharge data collection under drought conditions would increase modeling opportunities, thereby providing a better understanding of how low flow affects redd capacity.

### *Redd Capacity*

We reviewed the literature at the onset of our study to understand the problems others have encountered when estimating redd capacity. To our knowledge there are no peer-reviewed papers on this topic. Bjornn and Reiser (1991) reviewed unpublished data that clearly showed the potential for overestimating redd capacity when spawning area was based solely

on spawning substrate availability. They concluded that redd capacity depended on: the amount of suitable spawning substrate covered by water with acceptable depths and velocities for spawning (i.e., spawning area), and on the area required for a pair of spawning fish (i.e., spawning-area-per-redd).

We modified the Instream Flow Incremental Methodology (Bovee 1982) to estimate spawning area. Although widely applied by biologists, this method can grossly overestimate spawning area (Shrivell 1989). Using Shrivell (1989) for guidance, we made conservative estimates of spawning area by: 1) studying sites known to be used by spawners; 2) calculating spawning area based on the actual shape of the wetted spawning substrate patch rather than the rectangular shape of cells; and, 3) determining cell length using the location of redds or short stretches of habitat with relatively homogenous depths, velocities, substrate, and channel contours.

We used a relatively large value for spawning-area-per-redd (i.e., 70 m<sup>2</sup>) that was based on the highest redd density we observed. The space required for redd construction probably varies in response to stream size, spawn timing, and spawner density. For comparison, Swan (1989) reported spatial requirements ranging from 21.7 to 75.2 m<sup>2</sup>/redd. Burner (1951) proposed that female fall chinook salmon require four times the area of a redd to spawn, which equates to 68 m<sup>2</sup> using the redd surface area of 17 m<sup>2</sup> reported by Chapman et al. (1986). Using 70 m<sup>2</sup> added an additional measure of conservatism to our estimates of redd capacity.

We developed our method for estimating redd capacity to accomplish two objectives. The results obtained for the first objective indicate that redd capacity for the upper and lower reaches of the Snake River ranges from 2,466 to 2,570 under the stable flow regime. The actual carrying capacity of the Snake River for fall chinook salmon redds (or the "best estimate") might be higher because our method was conservative. For example, the estimates of redd capacity would have ranged from 7,875 to 8,283 if we divided spawning area by the 21.7 m<sup>2</sup> per redd reported by Swan (1989) instead of 70 m<sup>2</sup>.

### **Management Implications**

In light of our redd capacity estimates, we believe that the Snake River can support the 1,250 redds needed to remove Snake River fall chinook salmon from the list of federally protected species. The lowest of the three estimates, 2,466, is

roughly twice the de-listing criteria of 1,250 redds. We acknowledge that the 95% lower confidence limits on our redd capacity estimates show that redd capacity could be as low as 1,007 to 1,149. A stock-recruitment analysis (Ricker 1975) conducted with empirical data collected as spawner escapement increases will be the only way to confirm redd capacity, and to determine if the recovery goal is achievable. Other recovery measures such as spawning gravel enhancement might be necessary if recruitment efficiency is affected by density dependent factors before the recovery goal is attained.



### **Acknowledgements**

Many employees of the U. S. Fish and Wildlife Service's Idaho Fishery Resource Office and the Idaho Power Company collected and processed data. R. Taylor of the U. S. Forest Service was the project's professional surveyor. D. Geist collaborated on early drafts. T. Burton and an anonymous reviewer improved the manuscript. Funding was provided by the U. S. Fish and Wildlife Service Lower Snake River Compensation Plan and by the rate payers of the Bonneville Power Administration through Contracts DE-AI79-91BP21708 and DE-AC06-76RLO1830 administered by D. Docherty, M. Galloway, M. Beeman, and K. Tiffan.

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## CHAPTER FOUR

### Juvenile Life History of Wild Fall Chinook Salmon in the Snake and Clearwater Rivers\*

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\*Accepted for publication in the North American Journal of  
Fisheries Management

*Abstract.*—Dam construction in the 1950s and 1960s blocked passage to the historical spawning area of Snake River fall chinook salmon *Oncorhynchus tshawytscha*. We compiled water temperature data and collected juvenile fall chinook salmon in three present-day spawning areas from 1992 to 2000 to investigate the relation between water temperature and juvenile life history events. We used historical water temperature and the literature to depict juvenile life history in the historical spawning area. Water temperatures of the three present-day spawning areas differed significantly from winter through spring when eggs were incubating ( $P \leq 0.0001$ ), and during spring when juveniles were rearing and starting seaward movement ( $P \leq 0.0001$ ). Life stage timing generally proceeded earlier when water temperature was warmer than when it was cooler (fry emergence  $r^2 = 0.85$ ,  $N = 14$ ,  $P \leq 0.0001$ ; growth to parr size  $r^2 = 0.94$ ,  $N = 15$ ,  $P \leq 0.0001$ ; smolt migration  $r^2 = 0.93$ ,  $N = 14$ ,  $P \leq 0.0001$ ). The percentage of fish that overwintered in freshwater increased as water temperature during the spring decreased ( $r^2 = 0.40$ ,  $N = 12$ ,  $P = 0.02$ ). The historical spawning area was warmer than present-day spawning areas. Therefore, fall chinook salmon juvenile life history progressed on an earlier time schedule. We conclude that dam construction changed juvenile fall chinook salmon life history in the Snake River basin by shifting production to areas with relatively cool water temperatures and comparatively lower growth opportunity.

## Introduction

Construction of dams altered the freshwater habitat of anadromous salmonids in North America (e.g., Moffitt et al. 1982; Wunderlich et al. 1994; Kondolf et al. 1996; Dauble and Watson 1997; Dauble and Geist 2000). Dam construction can be a factor for population decline, especially if the historical spawning and rearing habitat were eliminated and the migration corridor was impounded. This was the case with Snake River fall chinook salmon *Oncorhynchus tshawytscha*, a stock that was listed as threatened under the Endangered Species Act (ESA) in 1992 (NMFS 1992).

Prior to 1963, the primary area for Snake River fall chinook salmon production was in a 49-km reach of river between Swan Falls Dam and Marsing, Idaho (Groves and Chandler 1999; hereafter, the Marsing reach of the Snake River; Figure 1). Fall chinook salmon were extirpated from the Marsing reach of the Snake River by Brownlee, Oxbow, and Hells Canyon dams (Figure 1). Lower Granite, Little Goose, Lower Monumental, and Ice Harbor dams (Figure 1) impounded the lower 224 km of the Snake River by 1975. This left a continuous 173-km reach of riverine habitat between Hells Canyon Dam and the upper end of Lower Granite Reservoir (Figure 1) for fall chinook salmon production. Spawners also had access to the lower reaches of the Imnaha, Salmon, Grande Ronde, and Clearwater rivers (Figure 1), which lack conclusive historical evidence of supporting the Snake River stock of fall chinook salmon.

From 1991 to 1999, annual redd searches were conducted in the Snake, Imnaha, Salmon, and Grande Ronde rivers, and in the Clearwater River basin (Idaho Power Company, Nez Perce Tribe, and U. S. Fish and Wildlife Service, unpublished data). A grand total of 1,867 fall chinook salmon redds was counted with an inter-annual range of 54 to 579. Timing of redd construction ranged from October to early December with a peak in early to mid November. Approximately 58% of the redds was counted in the Snake River, 27% in the lower Clearwater River, and 15% in the other areas combined.

Offspring of fall chinook salmon spawners make up the majority of wild fry and parr that inhabit the shorelines of the Snake and lower Clearwater rivers. From 1995 to 1997, genetic samples showed that 100% of the wild chinook salmon in the Snake River upstream of the Salmon River confluence (hereafter, the upper reach; Figure 1) were subyearling fall chinook salmon (Marshall et al. 2000; W. P. Connor, unpublished data). From

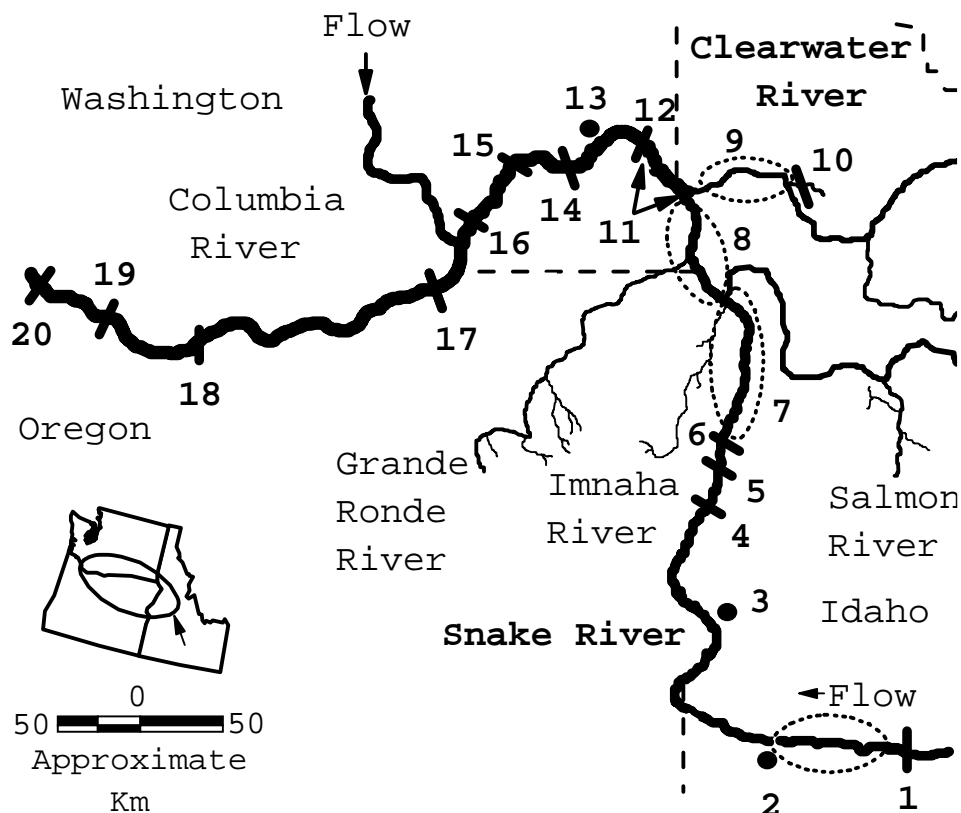


Figure 1.—Locations of the upper and lower reaches of the Snake River, the lower Clearwater River where adult fall chinook salmon spawn and their offspring were captured by using a beach seine (cross hatched ellipses), the historical spawning area between Swan Falls Dam and Marsing, Idaho (cross hatched ellipse), and other landmarks. The locations are as follows: 1 = Swan Falls Dam; 2 = Marsing, Idaho; 3 = Dipper trap (Krcma and Raleigh 1970); 4 = Brownlee Dam; 5 = Oxbow Dam; 6 = Hells Canyon Dam; 7 = Snake River upper reach; 8 = Snake River lower reach; 9 = Lower Clearwater River; 10 = Dworshak Dam; 11 = Lower Granite Reservoir; 12 = Lower Granite Dam (PIT-tag monitoring); 13 = Fyke nets (Mains and Smith 1964); 14 = Little Goose Dam (PIT-tag monitoring); 15 = Lower Monumental Dam (PIT-tag monitoring); 16 = Ice Harbor Dam; 17 = McNary Dam (PIT-tag monitoring); 18 = John Day Dam (PIT-tag monitoring); 19 = The Dalles Dam; 20 = Bonneville Dam (PIT-tag monitoring).



1993 to 1998, roughly 76% of the wild subyearling chinook salmon in the Snake River between the Salmon River confluence and the upper end of Lower Granite Reservoir (hereafter, the lower reach; Figure 1) was progeny of fall chinook salmon spawners (Connor et al. 2001a). The remaining 24% was offspring of spring and summer chinook salmon that spawn in low-order streams in the Snake River basin. These wild subyearling spring and summer chinook salmon dispersed long distances from natal streams into the Snake River where they reared and migrated seaward a little earlier than fall chinook salmon (Connor et al. 2001a, 2001b). Small amounts of data also suggest the presence of early rearing and seaward migrating subyearling spring chinook salmon in the lower Clearwater River (W. P. Connor, unpublished data). For simplicity, we refer to the wild subyearling chinook that inhabit present-day spawning areas as fall chinook salmon.

In this chapter, we describe and compare the water temperature regimes of the upper and lower reaches of the Snake River, and the lower Clearwater River. We describe and compare life history of juvenile fall chinook salmon in these three spawning areas. We test the relation between water temperature and juvenile life history events. We use historical water temperature data, growth opportunity theory (Metcalf and Thorpe 1990; Taylor 1990), and past studies to depict juvenile fall chinook salmon life history in the Marsing reach of the Snake prior to dam construction.

## **Methods**

Water temperature and life history data are presented by brood year. For example, mean water temperature from winter 1991 to spring 1992 is reported as water temperature in brood year 91. Young fall chinook salmon collected in spring of 1995, which were produced by spawning in 1994, are identified as being from brood year 94.

*Water temperature data.*—Data for brood years 91 to 98 were collected using hourly recording thermographs or standard U. S. Geological Survey temperature monitoring equipment stationed known distances (km) from each river mouth (abbreviated as rkm). Thermograph locations in the Snake River varied by year and flow level. Thermographs were stationed at rkm 383, rkm 369, rkm 346, rkm 325, and rkm 303 in the upper reach of the Snake River, and at rkm 290, rkm 287, rkm 274, rkm 265, and rkm 251 in the lower reach of the Snake River. Data were collected at rkm 35 (brood years 92 to 94) and rkm 19 (brood years 91 and 95 to 98)

in the lower Clearwater River. Data for the Marsing reach of the Snake River were collected at Swan Falls Dam for brood years 60 to 69 using a continuously recording thermograph.

Daily mean water temperature was calculated from thermograph output. Data for two or more thermographs in the Snake River were averaged within a reach to provide one daily mean water temperature value. Missing daily mean values were predicted by using ordinary least-squares regression (range of  $r^2$  values 0.93 to 0.99). For example, missing daily mean values were predicted for 10/15/68 to 10/22/68 based on a regression model fit using observed day of year (e.g., January 1 = 1) and daily mean water temperatures collected two weeks before 10/15/68 and two weeks after 10/22/68.

We calculated two water temperature indices from the daily mean water temperature data for comparison among spawning areas, and for regression analyses. Winter-spring (12/21 to 6/20) water temperature was used to index conditions during egg incubation and fry emergence. Mean spring (3/20 to 6/20) water temperature was used to index conditions during shoreline rearing and the onset of seaward migration.

*Life history.*—We sampled the upper reach of the Snake River from brood years 94 to 99, the lower reach of the Snake River from brood years 91 to 99, and the lower Clearwater River from brood years 92 to 94. We captured juvenile fall chinook salmon by using a beach seine (Connor et al. 1998). Sampling typically started in April soon after fry began emerging from the gravel, and was conducted 1 d/week at permanent stations within each spawning area. Once a majority of the fish were at least 60 mm fork length, we sampled additional stations in each spawning area for three consecutive weeks. We discontinued all sampling in June or July when the majority of fish had moved into Lower Granite Reservoir or to points further downstream.

We used the capture dates of fish under 46 mm fork length to describe fry emergence timing. We used the capture dates for fish over 45 mm fork length to describe the timing of growth to parr size. All capture dates were adjusted to Sunday's date the week of sampling to account for differences in day of sampling among the three spawning areas. For example, a capture date of 5/2/93 (Sunday) was reported for fry and parr collected from 5/4/93 to 5/6/93 (Tuesday to Thursday).

We inserted passive integrated transponders (PIT) tags (Prentice et al. 1990b) into parr 60 mm fork length and longer

(Connor et al. 1998). Tagged parr were released at the collection site after a 15-min recovery period. Some of the PIT-tagged fish were detected as smolts as they passed downstream in the juvenile bypass systems of dams equipped with PIT-tag monitors (Matthews et al. 1977; Prentice et al. 1990a; Figure 1). Operation schedules for the fish bypass systems varied by dam and year. Most of the detections were in the fish bypass systems of Lower Granite, Little Goose, and Lower Monumental dams operated from early April to early November, and at McNary Dam (Figure 1) operated from early April to early December.

We used the detection data collected at Lower Granite Dam, which is the first dam encountered en route to the Pacific Ocean, to represent smolt migration timing. We used the detection data collected at all dams equipped with PIT-tag monitoring equipment (Figure 1) to determine the annual percentage of tagged fish that overwintered in freshwater and migrated seaward the next spring. We calculated this percentage as: Number of fish released in year  $t$  that were last detected in year  $t+1$ , divided by the total number of fish released in year  $t$  that were detected in years  $t$  and  $t+1$ , multiplied by 100.

*Statistical analyses.*—We calculated grand mean winter–spring and spring water temperatures. For example, grand mean winter–spring water temperature was calculated as the mean of all the mean annual winter–spring water temperatures. We compared grand mean water temperatures by seasonal period among the three spawning areas using ANOVA ( $\alpha = 0.05$ ) with a randomized block design blocking on year with spawning area as the treatment. Tukey-type pair-wise comparisons ( $\alpha = 0.05$ ) were made to test for significant differences between grand means of two spawning areas.

We were unable sample all three spawning areas every year, therefore we did not statistically test for differences in life history. We generally compared life history based on the grand mean dates of fry emergence, growth to parr size, smolt migration, and the grand mean percentage of fish that overwintered in freshwater.

We used ordinary least-squares regression ( $\alpha = 0.05$ ) to test the relation between water temperature and life stage timing, and the percentage of fish that overwintered in freshwater and migrated seaward the next spring. We tested four null hypotheses: 1) fry emergence timing is not related to winter–spring water temperature; 2) timing of growth to parr

size is not related to spring water temperature; 3) smolt migration timing is not related to spring water temperature; and 4) the percentage of fish that overwintered in freshwater and migrated seaward the next spring is not related to spring water temperature.

## **Results**

### *Water Temperature Brood Years 91 to 98*

Winter-spring water temperatures (Table 1) differed significantly among the upper and lower reaches of the Snake River and the lower Clearwater River ( $P \leq 0.0001$ ). The upper reach of the Snake River was the warmest, followed by the lower reach of the Snake River, and then the lower Clearwater River (Table 1).

Spring water temperatures (Table 1) differed significantly among the three spawning areas ( $P \leq 0.0001$ ). The upper reach of the Snake River was the warmest, followed by the lower reach of the Snake River, and then the lower Clearwater River (Table 1).

### *Development through the Life Stages*

Fry emerged earliest in the upper reach of the Snake River, followed by the lower reach of the Snake River, and then the lower Clearwater River (Table 2). Fry generally emerged earlier when mean winter-spring water temperature was warmer than when it was cooler ( $P \leq 0.0001$ ; Figure 2).

Growth to parr size occurred earliest in the upper reach of the Snake River, and latest the lower Clearwater River (Table 3). Growth to parr size was generally earlier when mean spring water temperature was warmer than when it was cooler ( $P \leq 0.0001$ ; Figure 2).

We inserted PIT tags into a grand total of 13,605 parr. Of these, a grand total of 2,663 was detected as smolts as they passed Lower Granite Dam. Smolt migration timing was earliest for fish tagged in the upstream reach of the Snake River and latest for fish tagged in the Lower Clearwater River (Table 4). Smolts generally began seaward migration earlier when mean spring water temperature was warmer than when it was cooler ( $P \leq 0.0001$ ; Figure 2).

Table 1.—Mean water temperatures (°C) by seasonal period for the upper and lower reaches of the Snake River, and the lower Clearwater River, brood years 91 to 98. All three Grand means (°C+SE) within each seasonal period were significantly ( $\alpha = 0.05$ ) different.

Brood year	Winter–spring			Spring		
	Sneke River upper reach	Sneke River lower reach	lower Clearwater River	Sneke River upper reach	Sneke River lower reach	lower Clearwater River
91	8.9	9.0	7.9	12.7	12.9	11.1
92	7.6	7.1	5.7	11.8	11.0	8.6
93	8.0	7.8	6.3	12.0	11.8	9.2
94	8.2	7.6	5.7	11.8	10.9	8.2
95	8.7		6.1	12.7		8.2
96	8.7	7.9	6.8	12.4	11.2	8.9
97	8.3	8.1	7.3	12.0	11.5	9.9
98	8.5	7.5	6.3	12.3	10.6	8.2
Grand means	8.4+0.2	7.9+0.2	6.5+0.3	12.2+0.1	11.4+0.3	9.0+0.4

Table 2.—Wild fall chinook salmon fry (under 45 mm fork length) emergence timing given as Sunday's date for each week for the upper and lower reaches of the Snake River and the lower Clearwater River, brood years 91 to 99. Calendar dates are reported as the median; range; sample size. Grand means are reported as day of year  $\pm$  SE.

Brood year	Sneke River upper reach	Sneke River lower reach	lower Clearwater River
91		4/26;3/29-5/24;355	
92		5/16;4/4-6/20;199	6/27;6/27-7/4;18
93		5/15;4/3-6/5;440	6/5;4/24-6/26;54
94	4/23;4/2-5/21;117	4/30;4/2-6/4;257	6/18;4/2-7/2;90
95	4/28;4/14-5/5;14	5/5;4/14-6/23;268	
96	; ;1	5/4;4/20-6/29;114	
97	4/19;4/12-5/10;101	4/26;4/12-6/14;322	
98	5/2;4/4-5/23;97	5/2;4/4-6/27;278	
99	4/9;4/2-5/14;683	4/9;4/2-6/4;415	
Grand means	113 $\pm$ 4	122 $\pm$ 4	168 $\pm$ 6

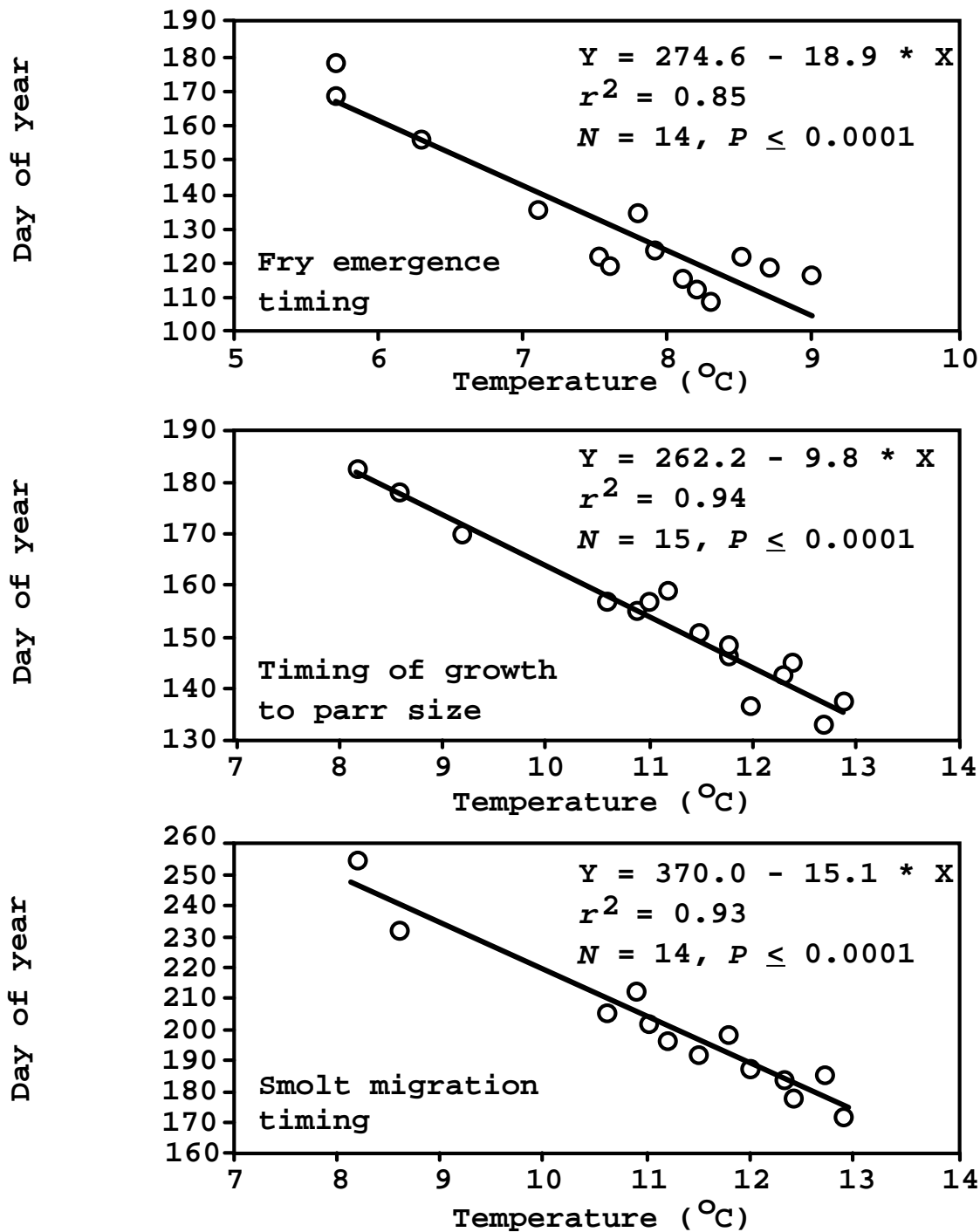


Figure 2.—Relations between fry emergence timing and winter-spring water temperature (Top), timing of growth to parr size and spring water temperature (Middle), and smolt migration timing and spring water temperature (Bottom), for wild fall chinook salmon in the upper and lower reaches of the Snake River and the lower Clearwater River. Data for the regressions are given in Tables 1, 2, 3, and 4.

Table 3.—Timing of wild fall chinook salmon growth to parr size (over 45 mm fork length) given as Sunday's date for each week for the upper and lower reaches of the Snake River and the lower Clearwater River, brood years 91 to 99. Calendar dates are reported as the median; range; sample size. Grand means are reported as day of year  $\pm$  SE.

Brood year	Snake River upper reach	Snake River lower reach	lower Clearwater River
91		5/17;3/29-6/7;1,765	
92		6/6;4/11-7/18;2,215	6/27;6/27-7/18;533
93		5/29;4/3-7/10;4,346	6/19;4/10-7/10;967
94	5/28;4/9-6/18;985	6/4;4/2-7/2;1,408	7/2;5/7-7/23;695
95	5/12;4/14-6/16;118	5/26;4/14-7/14;756	
96	5/25;4/20-6/15;119	6/8;4/20-7/13;938	
97	5/17;4/12-7/5;1,078	5/31;4/12-7/5;2,512	
98	5/23;4/11-6/27;1,493	6/6;4/4-7/11;1,647	
99	4/23;4/2-6/11;1,064	5/14;4/2-6/25;1,578	
Grand means	137 $\pm$ 5	150 $\pm$ 3	168 $\pm$ 6



Table 4.—Smolt migration timing at Lower Granite Dam for wild fall chinook salmon that were initially captured, PIT tagged, and released in the upper and lower reaches of the Snake River, and the lower Clearwater River, brood years 91 to 99. Calendar dates are reported as the median; range; sample size. Grand means are reported as day of year  $\pm$  SE.

Brood year	Snake River upper reach	Snake River lower reach	lower Clearwater River
91		6/20;5/4-7/21;39	
92		7/21;5/31-10/25;234	8/20;7/14-10/5;19
93		7/17;5/23-11/1;193	; ;1
94	7/18;6/4-10/24;203	8/1;6/2-10/26;238	9/12;7/2-10/30;30
95	7/4;5/20-7/25;19	7/22;5/17-10/31;126	
96	6/27;6/4-8/13;22	7/16;6/14-10/13;97	
97	7/7;5/19-8/21;173	7/11;5/29-10/19;380	
98	7/3;6/2-8/28;319	7/25;6/1-8/30;241	
99	6/27;5/6-7/18;72	7/2;5/18-10/28;257	
Grand means	186 $\pm$ 3	196 $\pm$ 4	244 $\pm$ 12

A grand total of 3,528 of the PIT-tagged parr was detected as smolts as they passed dams in the lower Snake and Columbia rivers. The percentage of fish that overwintered in freshwater and migrated seaward the next spring was lowest for the upper reach of the Snake River and highest for the lower Clearwater River (Table 5). The percentage of fish that overwintered in freshwater and migrated seaward the next spring generally increased as spring water temperature decreased ( $P = 0.02$ ; Figure 3).

#### *Water Temperature Brood Years 60 to 69*

On average, winter-spring water temperature for brood years 60 to 69 were 1 to 3°C warmer in the Marsing reach of the Snake River than in the three present-day spawning areas during brood years 91 to 98 (Tables 1 and 6). Mean spring water temperatures were also an average of 2 to 4°C warmer in the Marsing reach of the Snake River than in the three present day spawning areas (Tables 1 and 6).

Table 5.—The percentage of PIT-tagged wild fall chinook salmon from the upper and lower reaches of the Snake River, and the lower Clearwater River, that overwintered in freshwater and migrated seaward the next spring based on detection data collected at dams in the Snake and Columbia rivers, brood years 91 to 99. The total number of final detections is given in parentheses, and grand means are reported  $\pm$  SE.

Brood year	Snake River upper reach	Snake River lower reach	lower Clearwater River
91		4.4 (68)	
92		15.7 (351)	67.1 (70)
93		24.6 (334)	84.6 (26)
94	0.9 (328)	3.8 (364)	6.3 (48)
95	3.3 (30)	4.7 (169)	
96	0.0 (45)	18.5 (173)	
97	1.9 (324)	3.9 (693)	
98			
99	4.3 (139)	13.9 (366)	
Grand means	2.1 $\pm$ 0.8	11.2 $\pm$ 2.9	52.7 $\pm$ 23.7

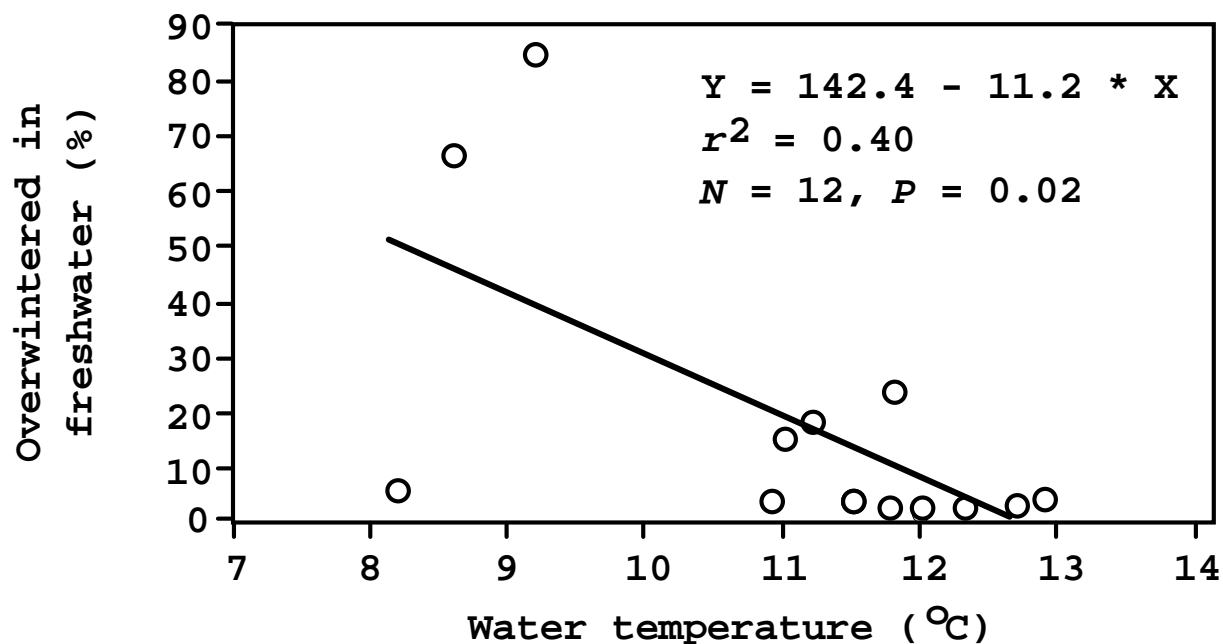


Figure 3.—The relation between spring water temperature and the percentage of PIT-tagged wild fall chinook salmon that overwintered in freshwater and migrated seaward the following spring based on detection data collected at dams in the Snake and Columbia rivers. Data for the regression are given in Tables 1 and 5.

Table 6.—Seasonal mean and grand mean  $\pm$  SE water temperatures ( $^{\circ}\text{C}$ ) for the Snake River measured at Swan Falls Dam, brood years 60 to 69.

Brood year										Grand mean
60	61	62	63	64	65	66	67	68	69	
Winter—spring										
9.5	10.2	9.0	9.7	8.5	9.1	9.7	9.9	10.0	10.2	9.6±0.2
Spring										
14.2	14.4	13.5	13.8	12.7	12.9	14.2	13.7	13.9	14.8	13.8±0.2

## Discussion

"As poikilotherms, young anadromous salmonids are strongly influenced by the temperatures of their habitat, and the extent of this influence is demonstrated in the life cycle of different species" (Brannon 1987). We found that life history of juvenile fall chinook salmon in the upper and lower reaches of the Snake River and the lower Clearwater River progressed on three separate but overlapping time schedules. We did not collect data to establish causal linkages between this finding and every factor that can affect life history variation in juvenile anadromous salmonids. Differences in the temperature regimes, however, offer the most plausible explanation for the life history variation we observed.

Fry emergence timing differed among present-day spawning areas largely because rate of egg development is positively correlated with water temperature, and because water temperature varied among spawning areas. Timing of growth to parr size, which was a crude measure of growth in fork length, varied among spawning areas partly because of fry emergence timing. The differences in water temperature among spawning areas also help explain variability in timing of growth to parr size because growth of young fall chinook salmon increases as water temperature increases within a range of 10.0 to 18.3°C provided that food is not limiting (Banks et al. 1971).

Water temperature also played a role on smolt migration timing. We suggest that there are two probable causes for this finding. Curet (1994) reported that juvenile fall chinook salmon reared along the shoreline of Lower Granite Reservoir later into the year when the water was cool, and that dispersal from the shoreline occurred when water temperature exceeded 18.0°C. Researchers have also suggested that fast growing chinook salmon progress from parr to smolt stage earlier in life than those that grow more slowly (Beckman and Dickhoff 1998; Connor et al. 2001b). Differences in timing of offshore movement and smoltification caused by water temperature would help explain the variability we observed in smolt migration timing.

Water temperature has been used by others to explain variability in anadromous salmonid life history. Metcalfe and Thorpe (1990) developed a growth opportunity index based on mean air temperature (used as a surrogate for water temperature) and photoperiod that explained 82% of the observed variability in age at smolting for wild Atlantic salmon *Salmo salar*. Juvenile

Atlantic salmon that reared in warmer stream reaches migrated seaward earlier than those from cooler stream reaches. Taylor (1990) analyzed data from 160 chinook salmon populations ranging from California to Alaska, and he showed that areas with low growth opportunity tended to produce juveniles that migrated seaward as yearlings.

The results in the present paper are consistent with those of Metcalfe and Thorpe (1990) and Taylor (1990). The warmest present-day spawning area, the upper reach of the Snake River, mainly fostered an "ocean-type" early life history (Healey 1991). Young fall chinook salmon emerged in spring, reared for two to three months, and then migrated seaward. The lower Clearwater River, which is the coolest present-day spawning area, sometimes produced juvenile chinook salmon with a "stream-type" early life history (Healey 1991). Fry emergence was in late spring and early summer. Many subyearling fall chinook salmon began seaward movement in summer and fall, overwintered in reservoirs, and then resumed seaward migration in spring.

Growth opportunity can be used as a basis for depicting juvenile fall chinook salmon life history in the Marsing reach of the Snake River. Historical redd surveys suggest that fall chinook salmon spawned at about the same time (if not earlier) as present-day spawners (Idaho Fish and Game, U. S. Fish and Wildlife Service, unpublished data). Therefore, the relatively warm Marsing reach of the Snake River would have fostered an ocean-type life history that progressed earlier than observed in present-day spawning areas during the 1990s. Studies by Krcma and Raleigh (1970) and Mains and Smith (1964) support this depiction, especially by comparison to the lower reach of the Snake River and the lower Clearwater River.

Krcma and Raleigh (1970) used a "migrant dipper" trap in 1962 and 1963 to capture offspring of adult fall chinook salmon that spawned in the Marsing reach of the Snake River in 1961 and 1962. The trap was located just upstream of Brownlee Reservoir (Figure 1), and it was operated daily from April through June. No fry were captured after mid-April in 1962, or after mid-May in 1963 (Krcma and Raleigh 1970). We captured fry in the lower reach of the Snake River from late May to early June, and in the lower Clearwater River from late June to early July. Approximately 98% of the juvenile fall chinook salmon population of the historical spawning area reached parr size and started migrating seaward by the end of May in both 1962 and 1963 (Krcma and Raleigh 1970). During the 1990s, an average of approximately 50% of the fish in the lower reach of the Snake

River and the lower Clearwater River had not grown to parr size or started moving seaward by the end of May.

Mains and Smith (1964) sampled juvenile anadromous salmonids in 1954 and 1955 using adjacent fyke nets that spanned the un-impounded Snake River at rkm 132 between the present locations of Lower Granite and Little Goose dams (Figure 1). Aging and genetic sampling was not conducted, but catch presumably included offspring of spring, summer, and fall chinook salmon that spawned throughout the Snake River basin in 1953 and 1954. Based on daily catch data, passage of the entire chinook salmon smolt run was complete by the end of June well before flow descended to base levels (Mains and Smith 1964). In the 1990s, an average of less than 50% of the smolts from the three main present-day fall chinook salmon spawning areas had passed Lower Granite Dam at rkm 173 by the end of June.

### **Management Implications**

We conclude that dam construction changed juvenile fall chinook salmon life history in the Snake River basin by shifting production to areas with relatively cooler water temperatures and comparatively lower growth opportunity. Consequently, smolt migrations do not begin until late spring and summer. Smolt passage in the lower Snake River reservoirs occurs after spring runoff has ended and summer water temperature reaches critical levels (Connor et al. 1998). Some young fall chinook salmon that survive in reservoirs over summer fail to reach the sea until they are yearlings.

The efficacy of the proposed Snake River fall chinook salmon recovery plan (NMFS 1995) relies in part on mitigation for dam-caused life history changes. Summer flow augmentation (Connor et al. 1998) and smolt transportation (Ward et al. 1997) are implemented annually to offset delays in seaward migration. Fishery managers need to know if summer flow augmentation increases downstream migration rate and survival of smolts in Lower Granite Reservoir, and if transportation of smolts from Lower Granite Dam increases smolt-to-adult return rates. Research is also needed to determine if egg incubation and growth could be accelerated by selective releases of upstream reservoir water.

## **Acknowledgments**

Employees of the U. S. Fish and Wildlife Service Idaho Fishery Resource Office, Nez Perce Tribe, Idaho Power Company, and U. S. Geological Survey collected data. Personnel of the Pacific States Marine Fisheries Commission maintained the PIT-tag data base. We express special thanks to B. Arnsberg, C. Eaton, P. Groves, A. Garcia, D. Marvin, R. Nelle, M. Pishl, D. Rondorf, J. Sneva, D. Steele, C. Stein, and K. Tiffan. Early drafts of the manuscript were reviewed by D. Bennett, P. Bigelow, J. Congleton, R. Graves, and K. Steinhorst. The editor, an associate editor, and two anonymous reviewers improved the manuscript. Initial funding was provided by the U. S. Fish and Wildlife Service Lower Snake River Compensation plan, and continued funding was provided by the rate payers of the Bonneville Power Administration through Contract Number DE-AI79-91BP21708 administered by D. Docherty, M. Galloway, and M. Beeman.



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## CHAPTER FIVE

Factors Affecting Downstream Migration Rate of Wild  
Subyearling Fall Chinook Salmon in the Snake River\*

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\*Submitted to the North American Journal of Fisheries Management

*Abstract.*—Summer flow augmentation is implemented annually to mitigate for the development of the hydropower system in the Snake River basin by increasing the downstream migration rate of wild subyearling fall chinook salmon *Oncorhynchus tshawytscha*. However, the efficacy of summer flow augmentation has been disputed. We studied some of the factors affecting downstream migration rate of wild subyearling fall chinook salmon in the Snake River. We then assessed the effects of summer flow augmentation on downstream migration rate, and on passage date at the first dam encountered by smolts during seaward migration. We used data collected on wild subyearling fall chinook salmon in the Snake River from 1995 to 2000 to fit year-by-year ordinary least-squares regression models. The predictor variables were release fork length (mm), release water temperature (°C), flow (m<sup>3</sup>/s), and distance traveled (km) in riverine habitat. Together these four variables explained from 62 to 86% of the observed variability in downstream migration rate (*N* range 119 to 560; all *P* values < 0.0001). On average, predicted downstream migration rates decreased from 0.1 to 0.2 km/d when the flow variable was reduced by the volume of water released for summer flow augmentation. We estimated that the average fish would have taken from 1 to 5 d longer to pass Lower Granite Dam without the aid of summer flow augmentation. The results in this paper suggest that summer flow augmentation increases downstream migration rate of fall chinook salmon smolts that are physiologically and behaviorally disposed to migrate seaward provided that the timing of flow augmentation corresponds to reservoir passage.

## Introduction

The migratory behavior of juvenile chinook salmon varies widely within the species. Ocean-type chinook salmon begin seaward migration as subyearlings after a few months of freshwater rearing (Healey 1991). Stream-type chinook salmon (Healey 1991) generally overwinter in their natal streams or larger-order streams, and then migrate seaward as yearlings the following spring (Chapman and Bjornn 1969; Bjornn 1971; Achord et al. 1996). Some young chinook salmon deviate from the early life history type of their parents and siblings by migrating one year earlier or later than normal (Connor et al. 2001a, 2001b, Chapter Four in this report).

Downstream movement of juvenile anadromous salmonids can include periods of dispersal and residency even after seaward migration is initiated (Smith 1982). Habitat alterations such as dams and reservoirs can result in seaward movement that is even more discontinuous. Venditti et al. (2000) found that 22% of the wild subyearling fall chinook salmon they radio tagged in a Snake River reservoir switched from downstream to upstream movement at some point in their migration, and that downstream migration rate declined as fish passed from faster flowing water in upper reservoir reaches to slower moving water in lower reservoir reaches. Venditti et al. (2000) attributed the upstream movement and reduction in migration rate they observed to decreased water velocity in the dam forebay.

The influence water velocity has on downstream migration rate of subyearling chinook salmon in reservoirs is disputed. Berggren and Filardo (1993) and Giorgi et al. (1997) studied the relation between stream discharge (a surrogate for water velocity, hereafter referred to as flow) and downstream movement of subyearling chinook salmon in the Columbia River. Berggren and Filardo (1993) concluded that increasing flow can mitigate for dam-caused passage delays. Giorgi et al. (1997) concluded that there is no evidence for a relation between downstream migration rate and flow.

The nature of the relation between downstream migration rate and water velocity (or a flow-based surrogate variable) is important to the efficacy of summer flow augmentation. Summer flow augmentation is made up of releases of water from Dworshak Reservoir and reservoirs upstream of Brownlee Dam (NMFS 1995; Connor et al. 1998; Figure 1). Summer flow augmentation is

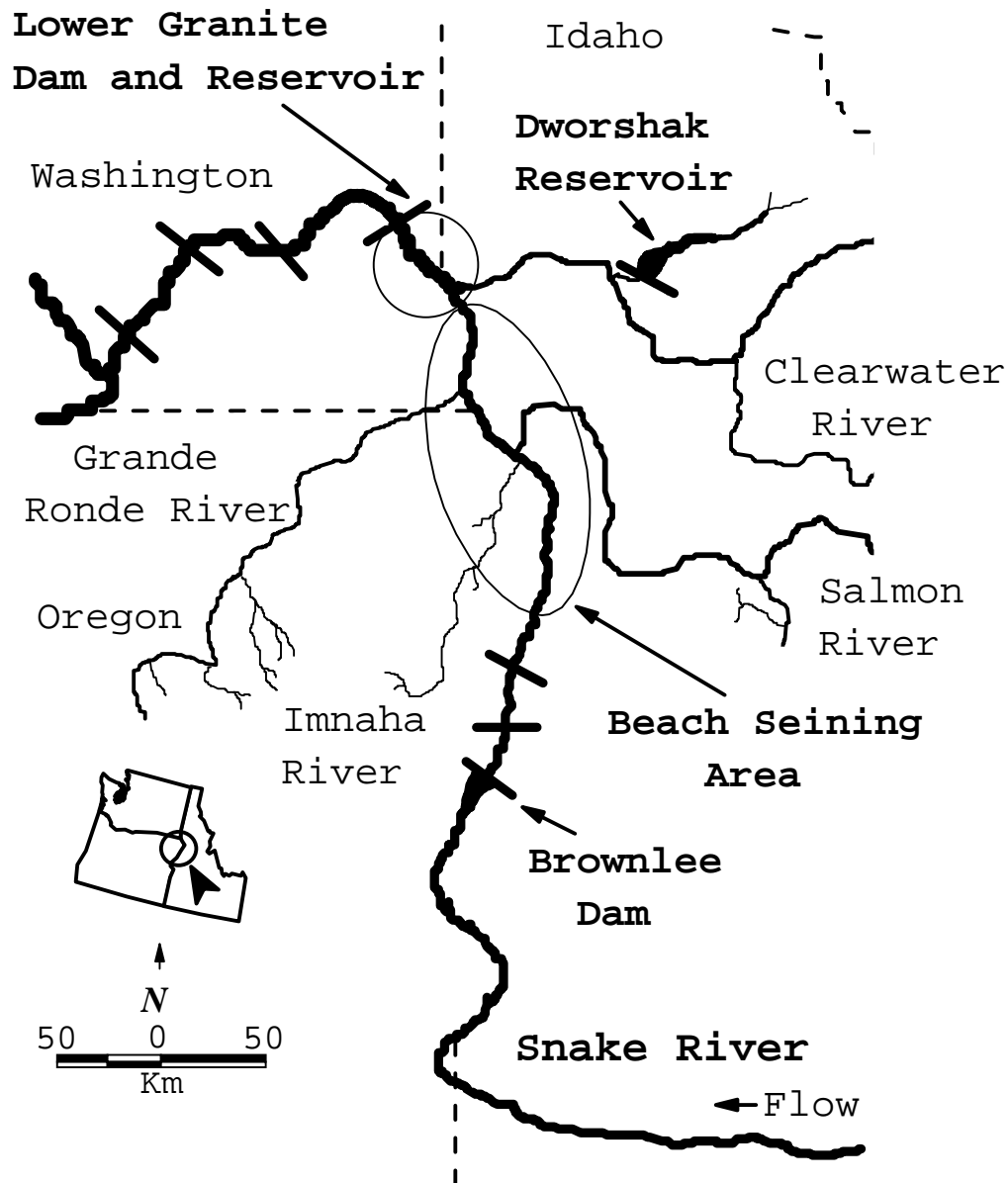


Figure 1.—The Snake River including the riverine habitat (cross-hatched elipse; rkm 224 to rkm 361) where adult fall chinook salmon spawn and their offspring were captured, tagged, and released to pass downstream in Lower Granite Reservoir (cross-hatched circle; rkm 173 to rkm 224) and past Lower Granite Dam (rkm 173), 1995 to 2000.

intended to help recover the Snake River stock of fall chinook salmon listed for protection under the Endangered Species Act (NMFS 1992) by increasing the migration rate of smolts as they pass downstream in impounded reaches of the Snake River including Lower Granite Reservoir (Figure 1). In this chapter, we identify some of the factors affecting migration rate of wild subyearling fall chinook salmon passing downstream in Lower Granite Reservoir, and then we estimate the effects of summer flow augmentation on downstream migration rate and passage date at Lower Granite Dam (Figure 1).

## Methods

*Data collection.*—Data collected from 1995 to 2000 were selected for our analyses for three reasons. Fall chinook salmon were captured and tagged throughout most of the free-flowing Snake River where adult fall chinook salmon spawn (Figure 1), flow varied widely among years, and wild subyearling fall chinook salmon composed approximately 76% of the catch (Connor et al. 2001a; W. P. Connor, unpublished data). The remaining 24% was composed of wild spring and summer chinook salmon that dispersed long distances from natal streams into the Snake River where they adopted an ocean-type life history similar to fall chinook salmon (Connor et al. 2001a, 2001b). For simplicity, we refer to all of the wild subyearling chinook salmon collected as fall chinook salmon.

Field personnel captured fall chinook salmon by using a beach seine (Connor et al. 1998). Sampling typically started in April soon after fry began emerging from the gravel, and was conducted 1 d/week at permanent stations within each spawning area. Once a majority of fish were at least 60 mm fork length, additional stations were sampled for three consecutive weeks. Sampling was discontinued in June or July when the majority of fish had moved into Lower Granite Reservoir or to points further downstream.

Passive integrated transponders (PIT) tags (Prentice et al. 1990b) were inserted into parr 60 mm fork length and longer (Connor et al. 1998). Tagged parr were released at the collection site after a 15-min recovery period. Some of the PIT-tagged fish were detected as smolts as they passed downstream in the juvenile bypass system of Lower Granite Dam (Matthews et al. 1977), which is equipped with PIT-tag monitors (Prentice et al. 1990a).



*Variables.*—The dependent variable for the analyses was downstream migration rate (km/d). Downstream migration rate was calculated on a fish-by-fish basis as:

distance (km) traveled to Lower Granite Dam divided by the number of days between release in the Snake River and detection at the dam.

The predictor variables for fitting the ordinary least-squares multiple regression model were:

release date = day of year (e.g., January 1 = 1) a fish was initially captured, tagged, and released back into the Snake River;

fork length = fork length (mm) measured when each fish was initially captured, tagged, and released back into the Snake River;

release temperature = water temperature (°C) measured at the sampling station using a hand-held thermometer when each fish was initially captured, tagged, and released back into the Snake River;

flow = mean stream discharge (m<sup>3</sup>/s) measured at Lower Granite Dam by U. S. Army Corps of Engineers personnel between the release and detection date of each fish at Lower Granite Dam; and

riverine distance = distance (km) traveled in the free-flowing Snake River before entering Lower Granite Reservoir.

*Model selection.*—On a year-by-year basis, a Pearson correlation coefficient ( $r$ ) was calculated to test for collinearity among the predictor variables. Predictor variables that were consistently correlated ( $r \geq 0.6$ ;  $P \leq 0.05$ ) were not entered into the same multiple regression model.

We log<sub>e</sub>-transformed downstream migration rate to improve linearity and remedy heteroscedasticity of residuals, and then we fit year-by-year multiple regression models from every combination of non-collinear predictor variables. Fit was compared among models based on Mallow's Cp scores (Dielman 1996), Akaike's information criteria (AIC)(Akaike 1973), and the coefficient of determination ( $R^2$ ). The final (i.e., best) regression model for each year had a Mallow's Cp score similar to the number of parameters, the lowest AIC value, the highest  $R^2$

value, and predictor variables with slope coefficients that differed significantly ( $t \geq 2.0$ ;  $P \leq 0.05$ ) from zero. Only the top three models were reported to shorten the results section.

We made year-by-year residual plots for each predictor variable in the final regression model as described for fork length in the following example.  $\log_e$ -transformed downstream migration rate was regressed against release temperature, flow, and riverine distance. The residuals from this regression were then plotted against fork length. A line was then fit to the residuals by regressing them against fork length. The resulting residual plots provided a better graphical representation of the relation between downstream migration rate and fork length because the variability in downstream migration rate attributable to the other predictor variables had been removed.

*Assessing flow augmentation.*—The effect of summer flow augmentation on downstream migration rate was assessed annually from 1995 to 2000. We predicted downstream migration rate for each PIT-tagged fall chinook salmon in a given year by inputting the observed values of the predictor variables, including flow, into the final multiple regression model for that year. We also predicted downstream migration rate for each PIT-tagged fish by inputting the observed value of each predictor variable, except for flow, into the final regression model. Flow, in this second prediction, was mean flow in Lower Granite Reservoir that would have occurred if summer flow augmentation had not been implemented (Appendices 1 and 2). Finally, we calculated two sets of passage dates at Lower Granite Dam for each fish by using the predicted downstream migration rates based on observed flows (i.e., with augmentation) and flows without augmentation.

## **Results**

A total of 6,134 fall chinook salmon were PIT tagged during the 6 years, of which 2,146 were detected passing Lower Granite Dam (Table 1). Downstream migration rates ranged from 0.4 to 44.8 km/d (Table 1). The ranges for the predictor variables were: release date 106 to 192; fork length 60 to 125 mm; release temperature 9.8 to 18.8°C; flow 825 to 5,609 m<sup>3</sup>/s; and riverine distance 1 to 141 km (Table 1).

Table 1.—Sample sizes (*N*) and ranges for variables collected on wild subyearling fall chinook salmon that were detected passing Lower Granite Dam after being PIT tagged and released in the Snake River, 1995 to 2000. Abbreviations: Rate = downstream migration rate (km/d); Date = release date (day of year); Fl = release fork length (mm); Degrees = release water temperature (°C); Flow = mean flow (m<sup>3</sup>/s) in Lower Granite Reservoir between release and detection at Lower Granite Dam; and Km = riverine distance (km) traveled to Lower Granite Dam.

Year	<i>N</i>	Rate	Date	Fl	Degrees	Flow	Km
1995	440	0.4-44.8	116-187	60-114	10.8-17.9	1187-3880	2-137
1996	145	0.4-43.8	107-192	60-111	9.8-18.6	893-5150	3-133
1997	119	0.5-21.6	128-191	60-108	11.3-18.4	1354-5158	1-133
1998	553	0.4-28.3	106-181	60-114	9.8-18.8	1265-5609	1-133
1999	560	0.7-35.0	118-189	60-112	10.0-18.1	1397-4872	3-141
2000	329	0.4-19.9	111-179	60-125	11.1-18.3	825-2696	3-133

### *Migration Rate Modeling*

Release date and flow were consistently correlated (range for  $r = 0.60$  to  $0.82$ ; all  $P$  values  $\leq 0.0001$ ). Therefore, these two variables were not entered into the same model. For all years, Mallows'  $C_p$  scores, AIC values, and coefficients of determination were best for models fit from fork length, release temperature, flow, and riverine distance (Table 2). The regression coefficients for each of these predictor variables (Table 3) differed significantly from zero ( $t \geq 2.0$ ;  $P < 0.05$ ). For the 6 years, 62.2 to 85.9% of the variability observed in downstream migration rate was accounted by these four predictor variables (Table 3). All model  $P$  values were  $\leq 0.0001$  (Table 3).

$\log_e$ -transformed downstream migration rate generally increased as each predictor variable increased as shown by the positive regression coefficients (Table 3). The slopes in the residual plots (Figures 2-7) also show that downstream migration rate increased with increases in fork length, release temperature, flow, and riverine distance.

### *Assessing Flow Augmentation*

During the years 1995 to 2000, from 62.9 to 90.0% of the PIT-tagged fall chinook salmon passed Lower Granite Dam when summer flow augmentation was being implemented (Table 4). On average, the mean flow experienced by fish that were exposed to summer flow augmentation increased by 110 to 167  $\text{m}^3/\text{s}$  or 3.9 to 6.7% (Table 4).

Summer flow augmentation increased annual mean downstream migration rate an average of 0.1 to 0.2 km/d based on predictions made for each fish regardless of its exposure history (Table 5). Predicted downstream migration rates for fish that were exposed to summer flow augmentation increased a maximum of 0.6 to 1.5 km/d (upper ranges in mean difference column of Table 5).

The average fish (regardless of exposure history) would have taken from 1 to 5 extra days to pass Lower Granite Dam if summer flow augmentation had not been implemented based on predicted downstream migration rates (Table 5). Calculated passage dates at Lower Granite Dam for fish that were exposed to summer flow augmentation were a maximum of 14 to 49 d earlier with than without summer flow augmentation (upper ranges in mean difference column of Table 5).

Table 2.—Mallow's Cp scores, Akaikes information criteria (AIC), and coefficients of determination ( $R^2$ ) used to compare the fit of year-by-year regression models describing the downstream migration rate of wild subyearling fall chinook salmon, 1995 to 2000. Abbreviations: Fl = release fork length (mm); Degrees = release water temperature ( $^{\circ}\text{C}$ ); Flow = mean flow ( $\text{m}^3/\text{s}$ ) in Lower Granite Reservoir between release and detection at Lower Granite Dam; and Km = riverine distance (km) traveled to Lower Granite Dam.

Year	C(p)	AIC	$R^2$	Variables in model
1995	5	-1127	0.859	Fl, Degrees, Flow, Km
	37	-1095	0.847	Fl, Flow, Km
	265	-922	0.773	Fl, Degrees, Flow
1996	5	-223	0.621	Fl, Degrees, Flow, Km
	14	-213	0.589	Fl, Flow, Km
	40	-191	0.516	Fl, Km
1997	5	-242	0.761	Fl, Degrees, Flow, Km
	12	-234	0.741	Fl, Flow, Km
	13	-233	0.738	Fl, Degrees, Flow
1998	5	-1282	0.723	Fl, Degrees, Flow, Km
	25	-1262	0.712	Fl, Flow, Km
	5	-1169	0.660	Date, Fl, Degrees, Km
1999	5	-1363	0.841	Fl, Degrees, Flow, Km
	88	-1285	0.817	Fl, Flow, Km
	181	-1209	0.790	Fl, Degrees, Flow
2000	5	-785	0.843	Fl, Degrees, Flow, Km
	10	-780	0.839	Fl, Flow, Km
	5	-575	0.701	Date, Fl, Degrees, Km

Table 3.—Year-by-year ordinary least-squares multiple regression models for describing the factors affecting migration rate of PIT-tagged wild subyearling fall chinook salmon passing downstream to Lower Granite Dam, 1995 to 2000. Abbreviations: B<sub>0</sub> = intercept; Fl = coefficient for release fork length (mm); Degrees = coefficient for release water temperature (°C); Flow = coefficient for mean flow (m<sup>3</sup>/s) in Lower Granite Reservoir between release and detection at Lower Granite Dam; and Km = coefficient riverine distance (km) traveled to Lower Granite Dam.

Regression coefficients							
B <sub>0</sub>	Fl	Degrees	Flow	Km	N	Model P	R <sup>2</sup>
<b>1995</b>							
-3.80528	0.01958	0.05498	0.00078	0.00595	440	≤ 0.0001	0.859
<b>1996</b>							
-3.79113	0.02437	0.09468	0.00032	0.01026	145	≤ 0.0001	0.622
<b>1997</b>							
-4.45163	0.03170	0.08540	0.00044	0.00422	119	≤ 0.0001	0.761
<b>1998</b>							
-3.19981	0.02501	0.06050	0.00031	0.00713	553	≤ 0.0001	0.723
<b>1999</b>							
-4.20790	0.02432	0.10308	0.00059	0.00526	560	≤ 0.0001	0.841
<b>2000</b>							
-4.06468	0.02910	0.02875	0.00107	0.00771	329	≤ 0.0001	0.843

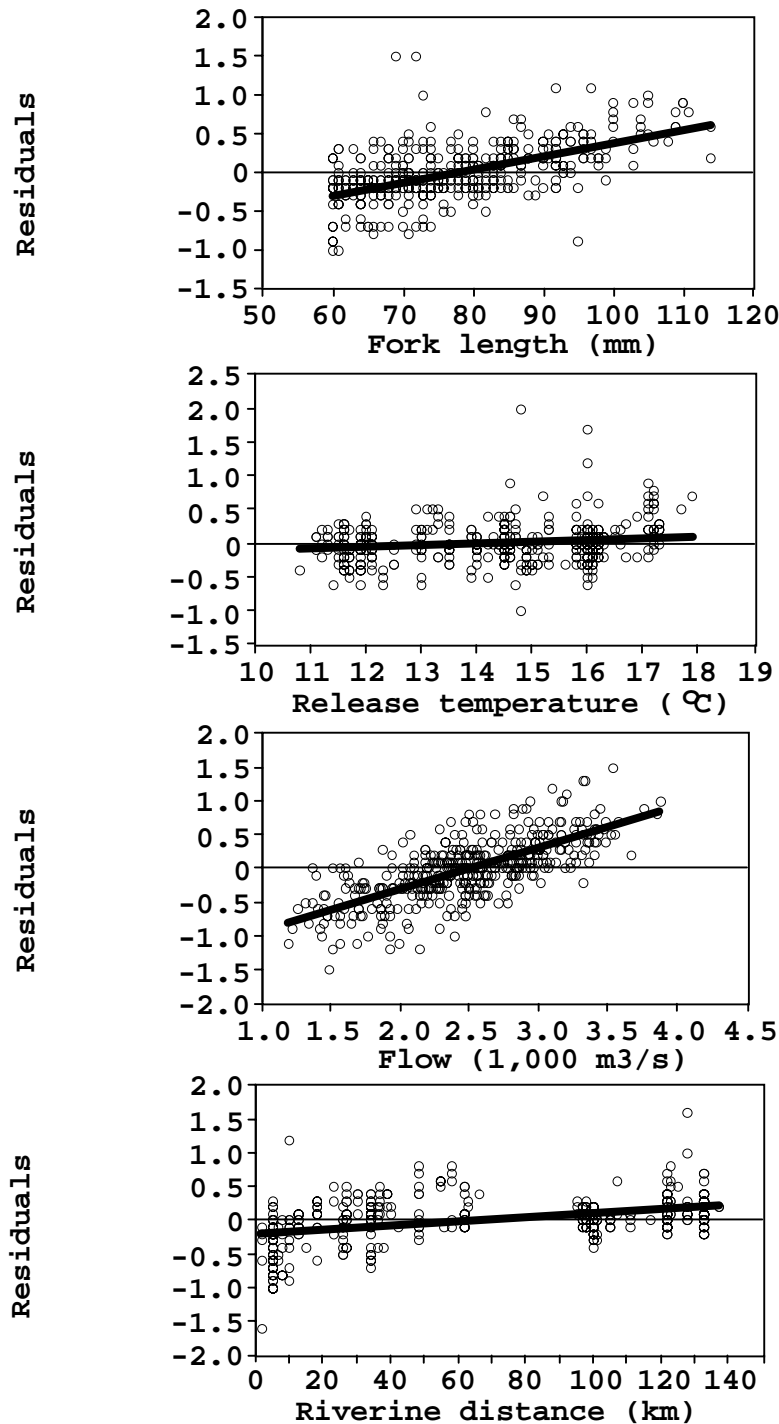


Figure 2.—Residual plots for each predictor variable in the 1995 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

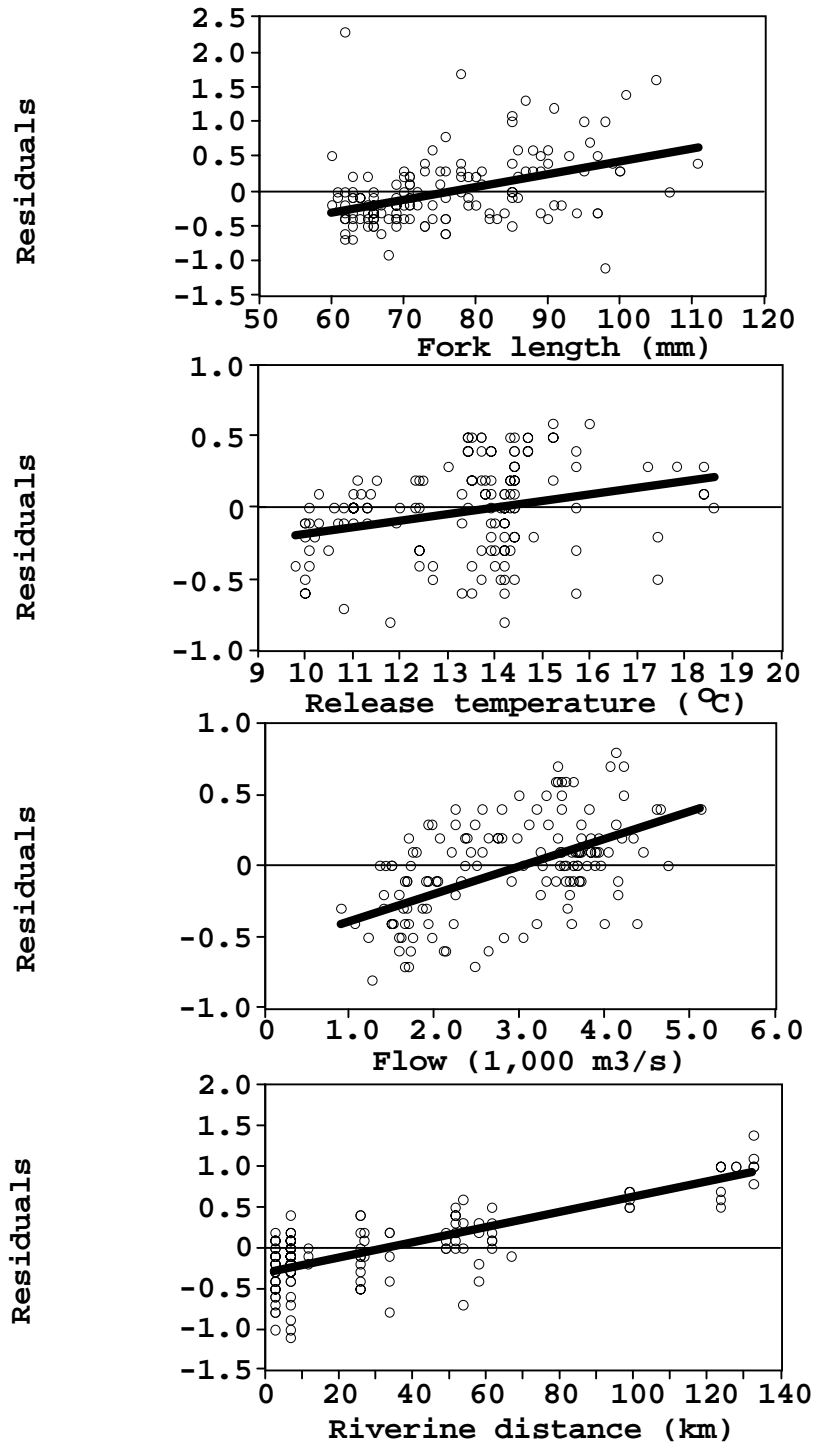


Figure 3.—Residual plots for each predictor variable in the 1996 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.



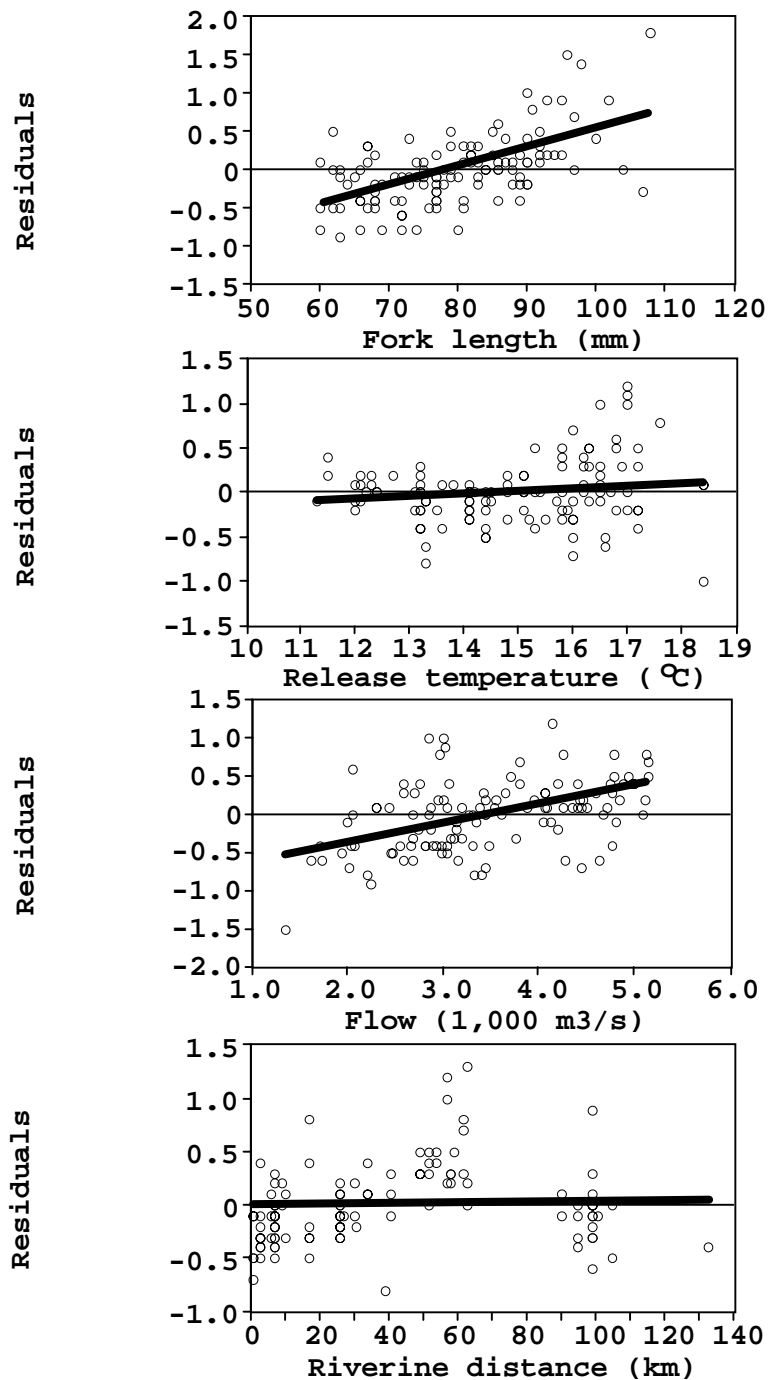


Figure 4.—Residual plots for each predictor variable in the 1997 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

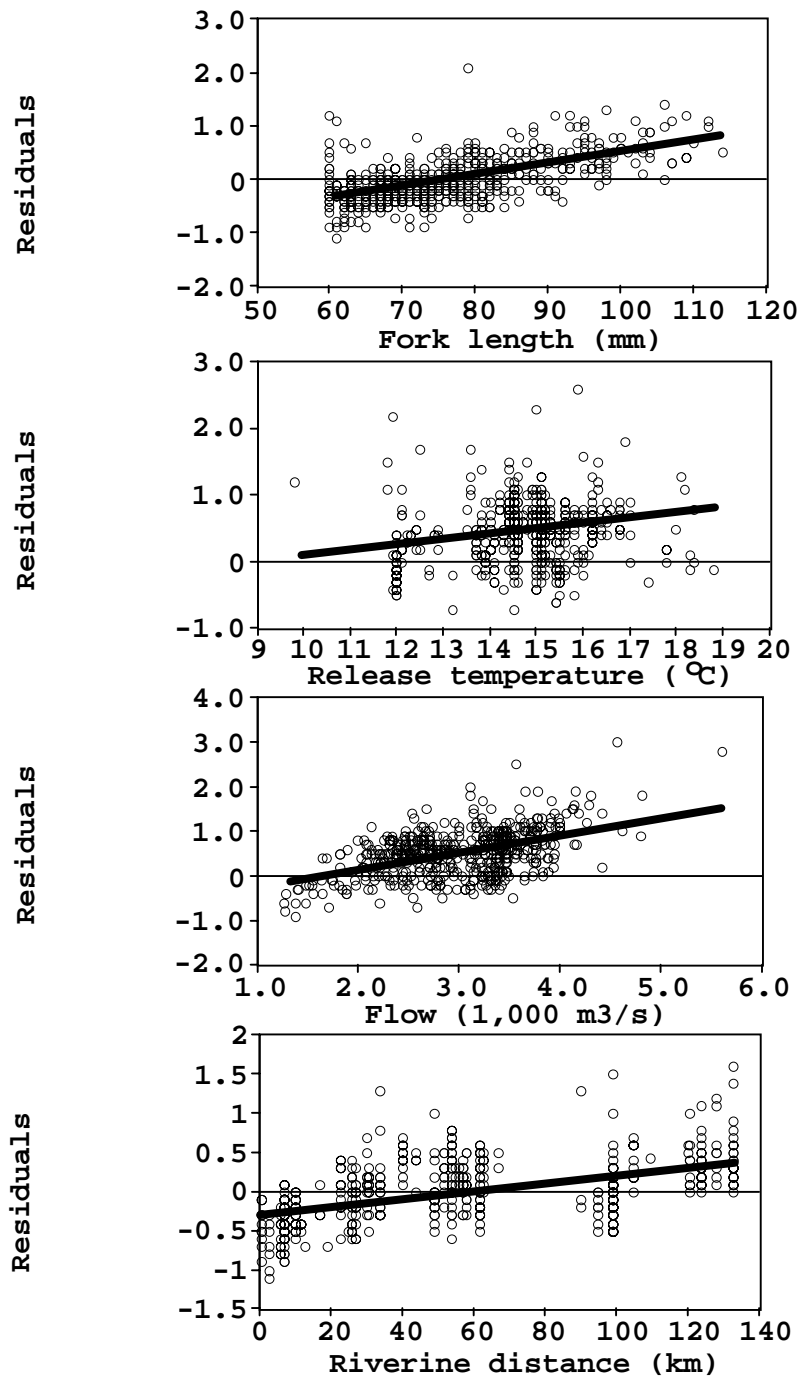


Figure 5.—Residual plots for each predictor variable in the 1998 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

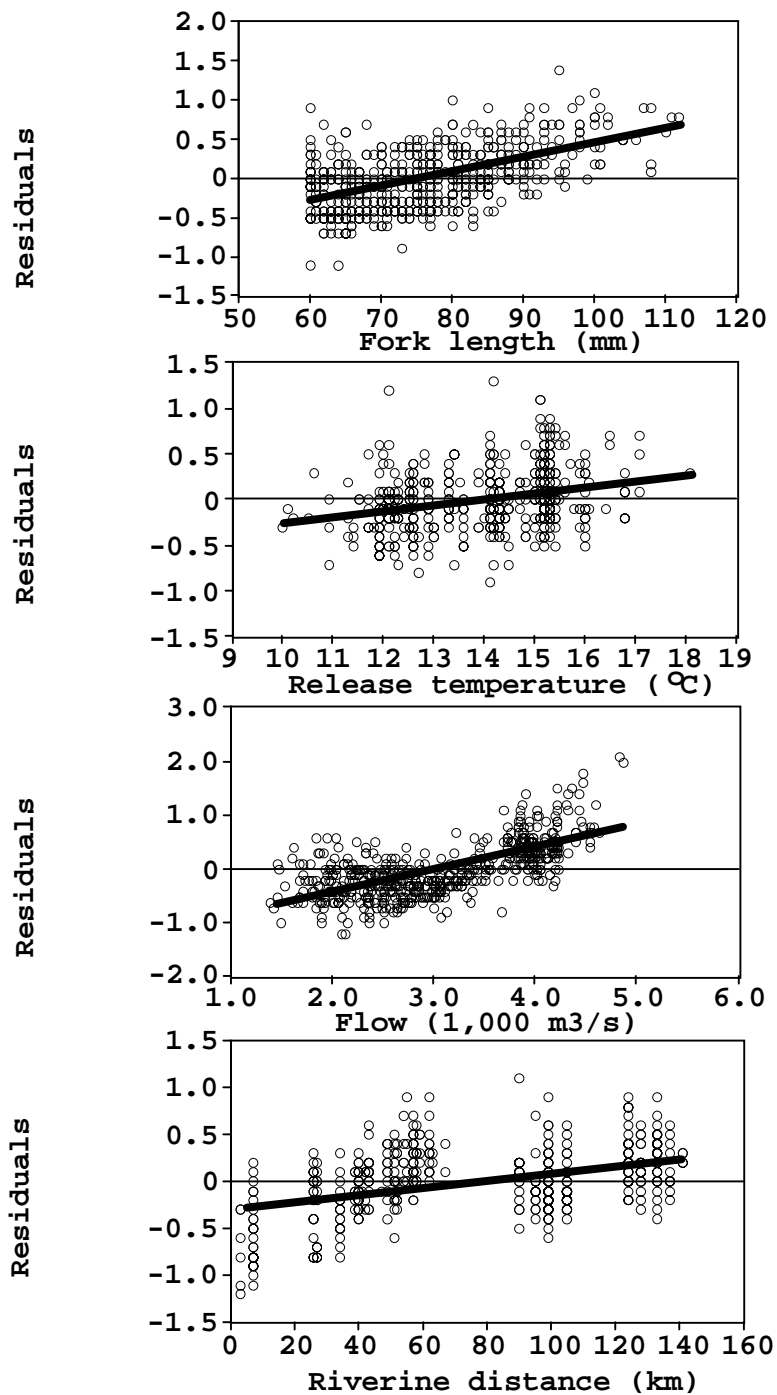


Figure 6.—Residual plots for each predictor variable in the 1999 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

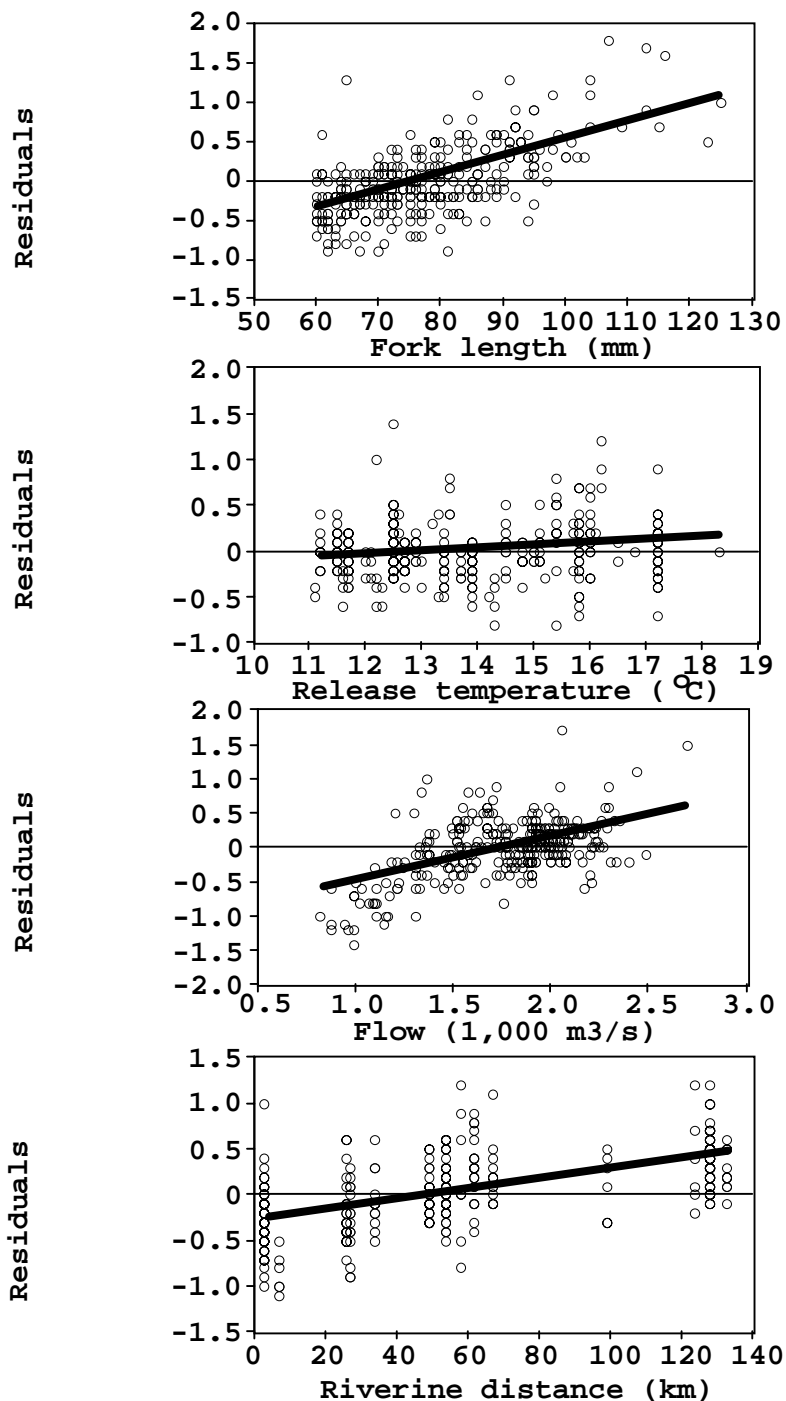


Figure 7.—Residual plots for each predictor variable in the 2000 final regression model. Residuals are from ordinary least-squares multiple regression models fit from all the predictor variables except the one on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

Table 4.—The percentage of PIT-tagged wild subyearling fall chinook salmon that passed Lower Granite Dam while summer flow augmentation was being implemented (Percentage exposed), and the mean flows ( $\text{m}^3/\text{s}$ ; range in parentheses) that these fish experienced (With) compared to those that would have occurred if summer flow augmentation had not been implemented (Without), 1995 to 2000. Also shown are the differences ( $\text{m}^3/\text{s}$  and %) with and without flow augmentation.

Year	N	Percentage exposed	<u>Mean Flow</u>		<u>Difference</u>	
			With	Without	$\text{m}^3/\text{s}$	%
1995	440	90.0	2,393 (1,187–3,230)	2,282 (905–3,227)	111	4.6
1996	145	85.5	2,676 (893–4,153)	2,540 (668–4,152)	136	5.1
1997	119	78.2	3,127 (1,354–4,672)	2,982 (911–4,669)	145	4.6
1998	553	81.0	2,810 (1,265–3,646)	2,700 (919–3,645)	110	3.9
1999	560	66.1	2,597 (1,397–3,924)	2,430 (921–3,921)	167	6.4
2000	329	62.9	1,607 (825–2,056)	1,500 (516–2,054)	107	6.7

Table 5.—Mean predicted migration rates (km/d; range in parentheses), and mean calculated passage dates at Lower Granite Dam (day of year; range in parentheses) for every PIT-tagged wild subyearling fall chinook salmon detected at Lower Granite Dam from 1995 to 2000. Abbreviations: With = observed mean flow conditions; Without = mean flow recalculated after reducing daily flows by the volume of water released for summer flow augmentation.

Mean migration rate		Mean difference	Mean passage date		Mean difference
With	Without		With	Without	
1995					
2.9 (0.5-13.5)	2.7 (0.4-13.5)	0.1 <sup>A</sup> (0-0.6)	208 (154-337)	213 (154-5 <sup>B</sup> )	6 <sup>C</sup> (0-39)
1996					
2.2 (0.7-9.1)	2.1 (0.6-9.1)	0.1 (0-0.5)	199 (155-256)	201 (155-267)	2 (0-14)
1997					
3.4 (0.6-14.5)	3.3 (0.6-14.5)	0.1 <sup>A</sup> (0-0.7)	194 (158-273)	197 (158-291)	2 <sup>C</sup> (0-18)
1998					
2.9 (0.7-8.9)	2.8 (0.7-8.9)	0.1 (0-0.6)	191 (157-259)	192 (157-259)	1 <sup>C</sup> (0-14)
1999					
4.7 (1.0-19.7)	4.5 (0.8-19.7)	0.1 <sup>A</sup> (0-1.1)	194 (153-269)	197 (153-278)	3 (0-26)
2000					
3.1 (0.6-17.4)	3.0 (0.4-17.4)	0.1 (0-1.5)	190 (139-285)	195 (139-322)	5 (0-49)

<sup>A</sup> Mean differences were calculated by averaging the observed differences for individual fish, not by subtracting the mean migration rate without flow augmentation from that with flow augmentation.

<sup>B</sup> 5 January, 1996

<sup>C</sup> Mean differences were calculated by averaging the observed differences for individual fish, not by subtracting the mean passage date with flow augmentation from that without flow augmentation.

## Discussion

Prior to becoming migratory, young Pacific salmon carry out activities that Hoar (1958) grouped into a small number of simple fixed behaviors. Downstream migration is a much less rigid behavior, and it is dependent upon the intensity and interaction of several directive factors (Hoar 1958). Hereafter, we broadly group these directive factors as being time based or environmentally based.

Release date is largely a time-based variable that has been used to index the progression of physiological processes (e.g., Berggren and Filardo 1993; Giorgi et al. 1997). It has also been used as a surrogate for flow to forecast smolt passage at Lower Granite Dam (Connor et al. 2000) because spring runoff generally declines throughout the late spring and early summer when fall chinook salmon are captured, tagged, and released. Flow and release date were highly correlated, thus we did not enter them into the same regression model. Only 2 of the top 18 regression models included release date as a predictor variable, and these two models were relatively poor predictors of downstream migration rate compared to those that included flow. These were important results because they strongly suggest that the flow fish experience as they migrate seaward has a greater influence than release date on downstream migration rate.

Fork length is a time-based variable that acts as a surrogate for level of smoltification (Berggren and Filardo 1993; Giorgi et al. 1997). In addition to becoming smolts at a critical size (e.g., Folmar and Dickhoff 1980; Wedemeyer et al. 1980), two other processes may have caused larger fall chinook salmon to migrate downstream faster than those that were smaller. Fall chinook salmon may have moved into faster deeper water as they grew (e.g., Chapman and Bjornn 1969; Lister and Genoe 1970; Everest and Chapman 1972), and they may have become more buoyant as they grew and became smolts (e.g., Wedemeyer 1996).

Release water temperature is an environmental variable that might stimulate fall chinook salmon to move offshore and downstream (Curet 1994; chapter four in this report). Keenleyside and Hoar (1954) studied juvenile chum *O. keta*, and coho *O. kisutch* in experimental troughs, and they found that fish switched from swimming upstream to downstream as temperature increased. Appetitive behavior (Hoar 1958) would help explain the switch to active downstream movement that Curet (1994), Keenleyside and Hoar (1954), and we observed. Fall chinook salmon released at warmer temperatures may have migrated

downstream faster than those released at cooler temperatures because they were searching for more comfortable rearing conditions.

Flow is an environmentally-based variable. Raymond (1968, 1979) was one of the first to focus on the relation between flow and downstream migration rate of chinook salmon smolts in the Snake River, and he concluded that downstream migration rate increased with flow. The variable flow is calculated by averaging daily stream discharge values. Stream discharge is the multiplicative product of mean water velocity, depth, and channel width. Therefore, stream discharge (hence flow) typically increases directly with water velocity. Fall chinook salmon that migrated seaward during periods of high flow may have migrated downstream faster than those released during lower flows because water velocity increased as flow increased.

Riverine distance is an environmentally-based variable that provided a second surrogate for water velocity. Water velocity in the Snake River is higher than in Lower Granite Reservoir (U. S. Army Corps of Engineers, unpublished data). Downstream migration rate may have increased with riverine distance traveled because fall chinook salmon released at upstream locations passed through long stretches of free-flowing river with high water velocities.

Thus far, we have discussed how single variables may have influenced downstream migration rate. All six year-by-year regression models presented in this paper, however, included both time-based and environmentally-based variables. These results are consistent with the hypothesis that downstream migration rate is a multivariate process influenced simultaneously by physiology, behavior, water temperature, and water velocity (Hoar 1958; Berggren and Filardo 1993). Based on these results, we conclude that migration rate will increase as flow increases provided that the physiological prerequisites for seaward migration are met, and fall chinook salmon are behaviorally disposed to move downstream.

### **Management Implications**

The results in this paper suggest that summer flow augmentation provides modest increases (mean differences 0.1 to 0.2 km/d) in migration rate of fall chinook salmon passing downstream in Lower Granite Reservoir. The increases in downstream migration rate, although small, translate to the average fish passing Lower Granite Dam from 1 to 5 d earlier



than without the aide of summer flow augmentation. The management implications of these findings can be interpreted two different ways depending on the reader's view on flow augmentation as a recovery tool.

Resource managers that believe the water used for summer flow augmentation should be allocated for other fishery and economic purposes might conclude that correlation between flow and downstream migration rate does not prove causation, and that our assessment of flow augmentation does not provide compelling evidence for successful mitigation. Advocates of summer flow augmentation might conclude that any increase in migration rate caused by summer flow augmentation is beneficial to population recovery, and that the results in this paper provide a basis for increasing the volume of water available for summer flow augmentation.

In our view, there is a need for a better understanding of the relation between flow and local water velocities where fall chinook salmon migrate in the Snake River and Lower Granite Reservoir, and on the response of smolts to changes in water velocity as they get closer to Lower Granite Dam. The existence of low water velocities in the forebay of Lower Granite Dam might disorient smolts and cause passage delays that are independent of flow (e.g., Venditti et al. 2000). The process by which fall chinook salmon locate fish bypass routes at dams may also be a chance event. The existence of these two phenomena in dam forebays would mask the true relation between downstream migration rate and flow.

It is also important to recognize that measurable differences in downstream migration rate are not the most important benefit of summer flow augmentation. Some of the water released for summer flow augmentation decreases and moderates temperature in Lower Granite Reservoir, which may in turn prevent growth reduction, impairment of smoltification, predation, and thermally induced mortality (Marine 1997; Connor et al. 1998). To truly understand the efficacy of summer flow augmentation, predictive models are needed to estimate the percentage of fall chinook salmon that survive to pass Lower Granite Dam where the majority of surviving smolts are collected and transported downstream (e.g., Ward et al. 1997).

## Acknowledgments

A. Garcia, D. Steele, C. Eaton, R. Nelle, R. Waitt, C. Larsen, J. Bretz, S. Bradbury and many other employees of the U. S. Fish and Wildlife Service's Idaho Fishery Resource Office collected data. The staff at the Fish Passage Center of the Columbia River Basin Fish and Wildlife Authority provided flow data. C. Stein, D. Marvin, and the staff at the Pacific States Marine Fisheries Commission managed the PIT-tag data base. D. Bennett, P. Bigelow, and J. Congleton reviewed drafts of the manuscript. The editor, an associate editor, and three anonymous reviewers improved the manuscript. Funding was the rate payers of the Bonneville Power Administration through Contract Number DE-AI79-91BP21708 administered by D. Docherty, M. Galloway, D. Praest, M. Beeman, and K. Tiffan.

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Appendix 1.—Mean daily flows (m<sup>3</sup>/s) in Lower Granite Reservoir with and without summer flow augmentation, 1995 to 1997.

Date	1995		1996		1997	
	With	Without	With	Without	With	Without
01-Jul	2359	2280	2062	2050	2846	2761
02-Jul	2271	2192	2345	2090	3036	2951
03-Jul	2147	2067	2588	2481	2699	2614
04-Jul	2628	2549	2317	2342	2546	2390
05-Jul	2322	2243	2271	2186	2433	2266
06-Jul	2101	2022	2212	2090	2322	2121
07-Jul	1999	1920	1948	1948	2087	1909
08-Jul	1974	1895	1722	1532	1999	1844
09-Jul	1923	1844	1759	1597	1931	1725
10-Jul	1991	1912	1521	1464	2065	1818
11-Jul	1917	1838	1422	1277	1963	1770
12-Jul	1954	1875	1504	1269	1634	1492
13-Jul	1889	1810	1478	1212	1606	1464
14-Jul	1866	1787	1441	1314	1532	1362
15-Jul	1609	1529	1354	1022	1770	1464
16-Jul	1543	1464	1566	1008	1829	1260
17-Jul	1453	1226	1538	1028	1821	1229
18-Jul	1560	1272	1436	1020	1807	1212
19-Jul	1473	1178	1374	1003	1815	1218
20-Jul	1410	1059	1272	918	1940	1340
21-Jul	1391	966	1260	903	1764	1150
22-Jul	1402	977	1189	804	1778	1161
23-Jul	1277	852	1218	807	1776	1150
24-Jul	1306	878	1172	756	1733	1099
25-Jul	1419	994	1195	719	1742	1104
26-Jul	1436	1014	1141	615	1660	1017
27-Jul	1419	994	1113	612	1637	991
28-Jul	1337	903	1147	620	1736	1087
29-Jul	1422	991	1045	572	1662	1011
30-Jul	1300	869	1102	668	1640	988
31-Jul	1320	898	1034	586	1682	1034
01-Aug	1303	847	1062	600	1589	835
02-Aug	1314	847	1099	640	1594	838
03-Aug	1246	776	1070	617	1626	858
04-Aug	1263	793	1037	592	1580	833

Appendix 1.-(Continued)

05-Aug	1255	784	1022	572	1492	756
06-Aug	1170	697	1028	592	1526	790
07-Aug	1218	745	1005	530	1535	804
08-Aug	1184	716	994	552	1467	745
09-Aug	1164	694	943	484	1464	739
10-Aug	1184	711	969	561	1487	762
11-Aug	940	467	997	564	1391	663
12-Aug	986	513	937	490	1470	742
13-Aug	940	467	957	532	1430	702
14-Aug	963	487	906	413	1467	742
15-Aug	986	496	1102	524	1388	671
16-Aug	949	464	1209	527	1289	739
17-Aug	1000	518	1150	464	1229	677
18-Aug	1039	561	1155	464	1212	660
19-Aug	1056	572	1167	510	1192	637
20-Aug	983	496	1099	459	1136	578
21-Aug	954	464	1087	459	1238	683
22-Aug	983	490	1104	484	1155	600
23-Aug	957	467	1133	507	1170	615
24-Aug	988	496	1068	471	1204	663
25-Aug	1003	513	1048	450	1209	671
26-Aug	946	456	1039	419	1221	671
27-Aug	949	459	1110	473	1161	685
28-Aug	875	382	1087	484	980	671
29-Aug	1005	513	1124	459	935	719
30-Aug	915	422	1167	411	841	626
31-Aug	969	473	1243	586	835	620

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Appendix 2.—Mean daily flows (m<sup>3</sup>/s) in Lower Granite Reservoir with and without summer flow augmentation, 1998 to 2000.

Date	1998		1999		2000	
	With	Without	With	Without	With	Without
01-Jul	2195	2138	2336	2243	1020	892
02-Jul	2212	2127	2212	2050	952	790
03-Jul	2251	2130	1931	1863	1014	835
04-Jul	2419	2283	1832	1702	977	816
05-Jul	2274	2116	1699	1594	1020	677
06-Jul	2065	1957	1685	1546	1090	773
07-Jul	1960	1844	1563	1427	1121	793
08-Jul	1827	1592	1546	1385	1059	552
09-Jul	1801	1515	1648	1458	1246	753
10-Jul	1778	1436	1563	1357	1198	583
11-Jul	1866	1385	1509	1269	1204	612
12-Jul	1892	1504	1532	1294	1274	572
13-Jul	1745	1087	1447	1136	1280	600
14-Jul	1812	1198	1529	1184	1229	513
15-Jul	1759	1164	1507	1172	1184	561
16-Jul	1651	1073	1507	1212	1161	501
17-Jul	1583	971	1475	1136	1187	507
18-Jul	1555	830	1541	1238	1087	524
19-Jul	1549	844	1501	991	1073	470
20-Jul	1577	881	1546	988	1099	504
21-Jul	1521	739	1456	954	1096	490
22-Jul	1535	719	1453	912	1028	450
23-Jul	1549	714	1456	895	1028	541
24-Jul	1512	688	1376	847	1005	382
25-Jul	1481	685	1354	824	1051	399
26-Jul	1444	646	1345	787	1076	467
27-Jul	1521	657	1314	762	1042	416
28-Jul	1529	762	1308	824	1031	515
29-Jul	1410	615	1257	685	860	436
30-Jul	1453	666	1263	671	643	530
30-Jul	1453	666	1263	671	643	530
31-Jul	1439	649	1368	634	855	453
01-Aug	1450	830	1357	617	833	408
02-Aug	954	765	1382	632	864	428
03-Aug	963	612	1323	615	784	402

Appendix 2--(Continued)

04-Aug	1283	705	1303	702	748	337
05-Aug	1167	586	1266	660	833	413
06-Aug	1201	634	1175	615	776	360
07-Aug	1065	592	1181	640	759	351
08-Aug	1107	671	1198	753	745	354
09-Aug	943	436	1116	555	733	326
10-Aug	1065	510	1141	671	813	362
11-Aug	1045	484	1054	600	813	377
12-Aug	1104	524	1028	547	733	280
13-Aug	1136	552	1164	694	787	368
14-Aug	1087	496	1028	697	773	362
15-Aug	1028	496	1090	702	750	297
16-Aug	960	524	1073	657	753	261
17-Aug	827	396	1170	711	799	365
18-Aug	954	445	1022	595	767	252
19-Aug	974	413	1025	578	858	408
20-Aug	1065	566	1070	544	787	354
21-Aug	932	521	1051	637	787	391
22-Aug	787	487	906	538	649	329
23-Aug	716	498	898	462	677	365
24-Aug	719	490	997	569	691	354
25-Aug	688	487	892	487	671	331
26-Aug	683	552	960	569	685	428
27-Aug	575	462	901	467	583	360
28-Aug	617	402	912	583	677	354
29-Aug	697	544	827	527	566	362
30-Aug	592	541	810	552	513	346
31-Aug	507	334	782	476	518	368

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## CHAPTER SIX

### Factors Affecting Survival of Wild Subyearling Fall Chinook Salmon in the Snake River\*

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*Abstract.*—Summer flow augmentation is implemented annually to mitigate for the development of the hydropower system in the Snake River basin by increasing the survival of wild subyearling fall chinook salmon *Oncorhynchus tshawytscha*. However, the efficacy of summer flow augmentation has been disputed. We studied some of the factors affecting survival of wild subyearling fall chinook salmon from release in the free-flowing Snake River to the tailrace of the first dam encountered by smolts en route to the sea. We then assessed the effects of summer flow augmentation on survival to the tailrace of this dam. We tagged and released a total of 5,030 wild juvenile fall chinook salmon in the free-flowing Snake River from 1998 to 2000. We separated these tagged fish into four sequential within-year release groups termed cohorts ( $N = 12$ ). Survival probability estimates to the tailrace of the dam for the 12 cohorts when summer flow augmentation was implemented ranged from  $35.7 \pm 4.3\%$  to  $87.7 \pm 4.6\%$ . We fit an ordinary least-squares multiple regression model from indices of flow and temperature that explained 92.3% ( $N = 12$ ;  $P \leq 0.0001$ ) of the observed variability in cohort survival. Survival generally increased with flow and decreased with temperature. We used the regression model to predict cohort survival for flow and temperature conditions observed when summer flow augmentation was implemented, and for the flow and temperature conditions that might have occurred if summer flow augmentation had not been implemented. Survival of all 12 cohorts was predicted to be higher when flow was augmented, than when flow was not augmented, because summer flow augmentation increased the flow levels and decreased the temperatures smolts were exposed to during seaward migration. We conclude that summer flow augmentation increases the survival of fall chinook salmon smolts.

## Introduction

Survival of chinook salmon smolts during seaward migration is affected by biotic factors, which in turn are controlled by the physical environment. Researchers have proposed that stream flow and temperature act together to influence survival of chinook salmon smolts (Kjelson et al. 1982; Kjelson and Brandes 1989; Connor et al. 1998). Dams have altered the flow and water temperature regimes of rivers in the western U. S., thereby contributing to declines in abundance of many stocks of chinook salmon by reducing smolt survival (e.g., Raymond 1988; Yoshiyama et al. 1998).

Raymond (1979) was the first to estimate survival for yearling Snake River spring and summer chinook salmon smolts, and to relate a decline in survival over years to dam construction. From 1966 to 1968, Raymond (1979) estimated that survival from the Salmon River to Ice Harbor Dam (Figure 1) for yearling spring and summer chinook salmon smolts ranged from 85 to 95%. Between 1970 and 1975 Lower Monumental and Little Goose dams (Figure 1) were completed and smolt survival estimates to Ice Harbor Dam decreased to a range of 10 to 50% (Raymond 1979). Raymond (1979) concluded that during high flow years lethal levels of dissolved gases killed yearling spring and summer chinook salmon smolts, whereas in low flow years mortality resulted from low reservoir water velocities, delayed reservoir passage, predation, and passage via dam powerhouses.

Wild subyearling chinook salmon that pass downstream in the lower Snake River reservoirs from May to August include spring, summer and fall-run juveniles that are listed for protection under the Endangered Species Act (NMFS 1992). Wild fall chinook typically comprise the majority of the subyearling smolts that pass downstream during summer in the lower Snake River (Connor et al. 2001a). The minority is composed of wild spring and summer chinook that disperse long distances from natal streams into the Snake River where they adopt an ocean-type life history similar to fall chinook salmon (Connor et al. 2001a, 2001b). For simplicity, we refer to all of the wild subyearling chinook salmon that inhabit the shorelines of the Snake River as fall chinook salmon.

Dam construction changed juvenile fall chinook salmon life history in the Snake River basin by eliminating production in the relatively warmer water of the historical spawning area, thereby restricting spawning to less productive cooler reaches of river (Chapter Four in this report). This helps explain why present-day smolts migrate seaward during summer in contrast to their pre-dam counterparts that migrated seaward in late spring (Chapter Four in this report). Summer flow augmentation is intended to help recover the Snake River stock of fall chinook

salmon by mitigating dam-caused changes in life history timing (NMFS 1995).

Summer flow augmentation is made up of releases of water from Dworshak Reservoir and reservoirs upstream of Brownlee Dam (NMFS 1995; Connor et al. 1998; Figure 1). These releases increase flow and decrease water temperature in Lower Granite Reservoir (Connor et al. 1998; Figure 1). Summer flow augmentation increases the migration rate of fall chinook salmon passing downstream in Lower Granite Reservoir, and reduces the time smolts take to pass Lower Granite Dam (Figure 1) by an average of 1 to 5 d (Chapter Cive in this report).

Connor et al. (1998) concluded that summer flow augmentation also increases fall chinook salmon survival to Lower Granite Dam, and recommended that future studies should include sequential within-year releases of tagged fish and survival estimation using a mark-recapture approach. In this chapter, we estimate survival from release in the free-flowing Snake River to the tailrace of Lower Granite Dam using a mark-recapture approach. We identify some of the factors affecting survival, and then we assess the effect of summer flow augmentation on survival.

### **Methods**

*Data collection.*—We analyzed data collected on fall chinook salmon from 1998 to 2000. Data for these years were selected because sample sizes of tagged fall chinook salmon were large, and tagged fish were not handled as they passed Lower Granite Dam. Field personnel captured fall chinook salmon by using a beach seine (Connor et al. 1998). Sampling typically started in April soon after fry began emerging from the gravel, and was conducted 1 d per week at permanent stations within each spawning area. Once a majority of fish were at least 60 mm fork length, additional stations were sampled for three consecutive weeks. Sampling was discontinued in June or July when the majority of fish had moved into Lower Granite Reservoir or to points further downstream.

Passive integrated transponders (PIT) tags (Prentice et al. 1990b) were inserted into parr 60 mm fork length and longer (Connor et al. 1998). Tagged parr were released at the

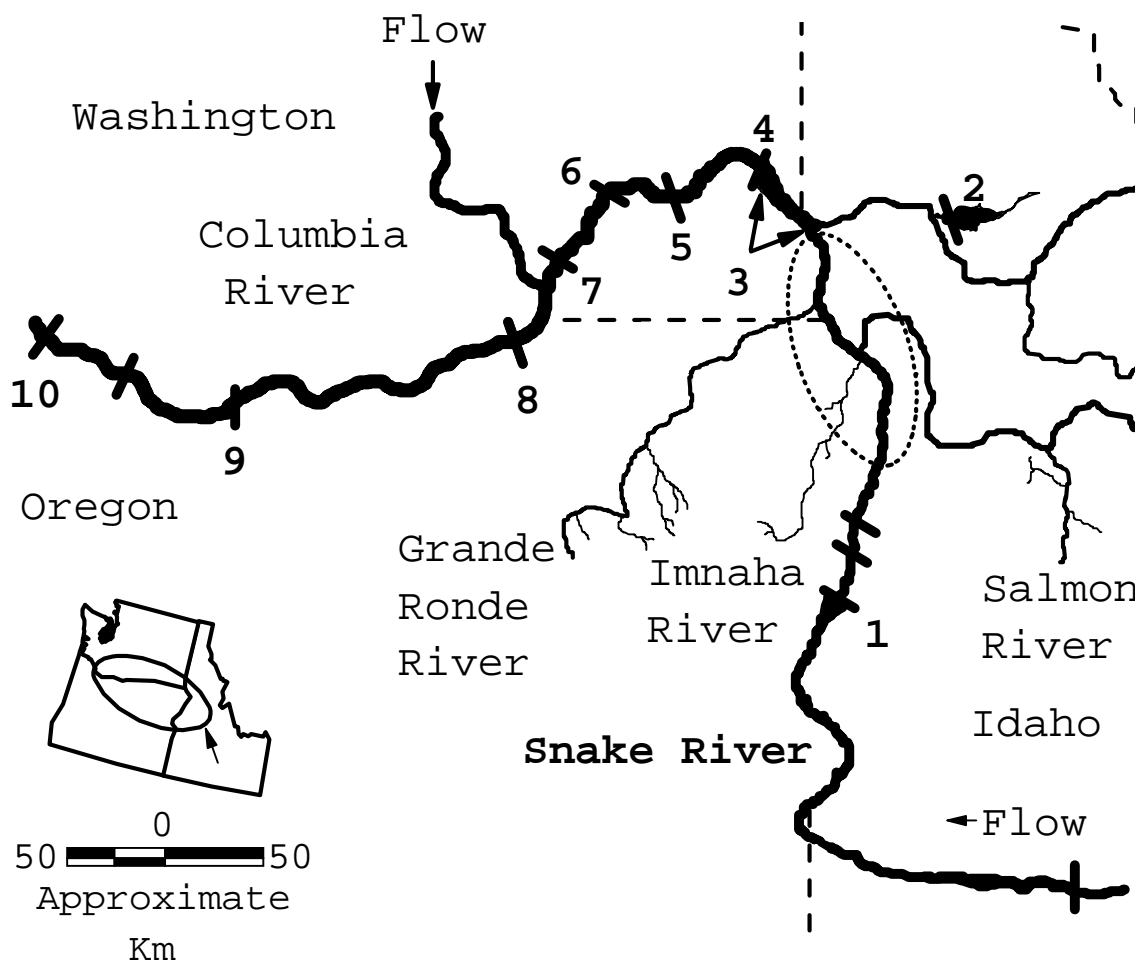


Figure 1.—Locations of the free-flowing Snake River where adult fall chinook salmon spawn and their offspring were captured by using a beach seine (cross hatched ellipse; rkm 224 to rkm 361) and other landmarks mentioned in the text. The locations are as follows: 1 = Brownlee Dam; 2 = Dworshak Reservoir; 3 = Lower Granite Reservoir; 4 = Lower Granite Reservoir (PIT-tag monitoring); 5 = Little Goose Dam (PIT-tag monitoring); 6 = Lower Monumental Dam (PIT-tag monitoring); 7 = Ice Harbor Dam; 8 = McNary Dam (PIT-tag monitoring); 9 = John Day Dam (PIT-tag monitoring), and 10 = Bonneville Dam (PIT-tag monitoring).

collection site after a 15-min recovery period. Some of the PIT-tagged fish were detected as smolts as they passed downstream in the juvenile bypass system of Lower Granite Dam (Matthews et al. 1977), which is equipped with PIT-tag monitors (Prentice et al. 1990a).

After detection at Lower Granite Dam, the PIT-tagged smolts were routed through flumes back to the river. Smolts then had to pass seven more dams (Figure 1) to reach the Pacific Ocean. Little Goose, Lower Monumental, McNary, John Day, and Bonneville dams (Figure 1) were also equipped with monitoring systems that recorded the passage of PIT-tagged smolts that used the bypass systems, and then routed the bypassed fish back to the river.

*Cohort survival.*—The first step in the analysis was to divide the annual samples of PIT-tagged fall chinook salmon into four sequential within-year release groups referred to as "cohorts." We divided the annual samples into cohorts based on estimated fry emergence dates. We estimated fry emergence date for each fish in two steps. First, the number of days since each PIT-tagged fish emerged from the gravel was calculated by subtracting 36 mm from its fork length measured at initial capture, and then dividing by the daily growth rate observed for recaptured PIT-tagged fish (range 0.9 to 1.3 mm/d; W. P. Connor, unpublished data). The 36-mm fork length for newly emergent fry was the mean of the observed minimum fork lengths. Second, emergence date was estimated for each fish by subtracting the estimated number of days since emergence from its date of initial capture, tagging, and release. We sorted the data in ascending order by estimated fry emergence date, and then divided it into four cohorts of approximately equal numbers of fish.

The single release-recapture model (Cormack 1964; Skalski et al. 1998) was used to estimate survival probability to the tailrace of Lower Granite Dam for each cohort. We insured that the single release-recapture model fit the data by using three assumption tests described by Burnham et al. (1987) and Skalski et al. (1998).

*Variables.*—Cohort survival was the dependent variable for the analysis. Variables for fitting the ordinary least-squares multiple regression model for predicting cohort survival were selected from the literature (Connor et al. 1998, 2000). The predictor variables were:

release date = median day of year fish from each cohort were captured, tagged, and released.

fork length = mean fork length (mm) at capture, tagging, and release for the fish of each cohort;

flow = a flow ( $\text{m}^3/\text{s}$ ) exposure index calculated as the mean flow measured at Lower Granite Dam by U. S. Army Corps of



Engineers personnel during the period when the majority of smolts from each cohort passed the dam; and,

temperature = a water temperature ( $^{\circ}\text{C}$ ) exposure index calculated as the mean temperature measured in the tailrace of Lower Granite Dam by U. S. Army Corps of Engineers personnel during the period when the majority of smolts from each cohort passed the dam.

To determine when the "majority" of smolts passed Lower Granite Dam, the PIT-tag detection data were used to calculate a passage date distribution for each cohort including mild outliers as described by Ott (1993; Figure 2). All but the mild outliers were considered to be in the majority. For example, the mean flow exposure index calculated based on the passage date distribution in Figure 2 would be the average of the mean daily flows measured in the tailrace of Lower Granite Dam between 6/17 and 8/16.

*Model selection.*—We calculated a Pearson correlation coefficient ( $r$ ) to test for collinearity among the predictor variables. Predictor variables that were correlated ( $r \geq 0.6$ ;  $P \leq 0.05$ ) were not entered into the same model.

We fit multiple regression models from every combination of non-collinear predictor variables. We compared fit among models based on Mallows'  $C_p$  scores (Dielman 1996), Akaike's information criteria (AIC) (Akaike 1973), and the coefficient of determination ( $R^2$ ). The final (i.e., best) regression model for each year had a Mallows'  $C_p$  score similar to the number of parameters, the lowest AIC value, the highest  $R^2$  value, and predictor variables with slope coefficients that differed significantly ( $t \geq 2.0$ ;  $P \leq 0.05$ ) from zero. Only the top three models were reported to shorten the results section.

We made residual plots for each predictor variable in the final regression model as described for flow in the following example. Estimated survival was regressed against temperature. The residuals from this regression were then plotted against flow. A line was then fit to the residuals by regressing them

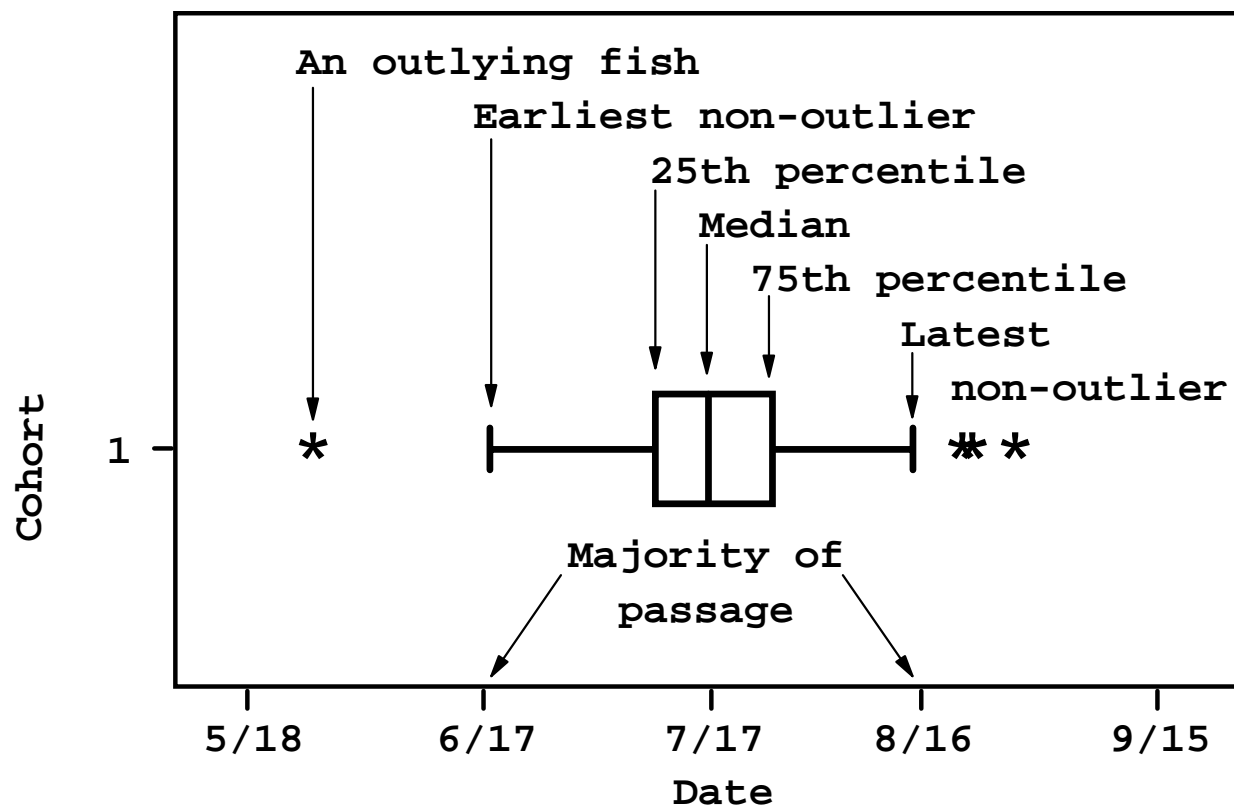


Figure 2.—An example of a passage date distribution for PIT-tagged wild subyearling fall chinook salmon at Lower Granite Dam including the time period that was used to represent the majority of passage for calculating flow and water temperature exposure indices.

against flow. The resulting residual plots provided a better graphical representation of the relation between survival and flow because the variability in survival attributable to temperature had been removed.

*Assessing summer flow augmentation.*—We assessed the effect of summer flow augmentation on cohort survival to the tailrace of Lower Granite Dam by comparing two predictions. First, we predicted cohort survival to the tailrace of Lower Granite Dam by inputting the observed mean flow and water temperature exposure indices for each cohort into the final regression model. Cohort survival was then predicted a second time by inputting mean flow and water temperature exposure indices into the final regression model that were recalculated to remove effects of summer flow augmentation.

The flow exposure index was recalculated after subtracting the daily volume of water released for summer flow augmentation (Appendix 1). The water temperature exposure index was recalculated using temperatures that were simulated for the tailrace of Lower Granite Dam under the flow conditions that would have occurred if summer flow augmentation had not been implemented (Appendix 2). Water temperatures were simulated using a one-dimensional heat budget model developed for the Snake River by the U. S. Environmental Protection Agency (Yearsley et al. 2001). Past model validation showed that daily mean water temperatures simulated for July and August were within an average of 1.1°C of those observed (Yearsley et al. 2001).

## **Results**

During the 3 years, 5,030 fall chinook salmon were captured, PIT tagged, and released along the free-flowing Snake River. Annual sample sizes of PIT-tagged fall chinook salmon were 2,060 in 1998, 1,761 in 1999, and 1,209 in 2000. The number of fall chinook salmon in the resulting 12 cohorts ranged from 302 to 515 (Table 1). Release dates, fork lengths, and water temperature exposure indices generally increased from cohort 1 to 4 (Table 1). Flow exposure indices and survival estimates decreased from cohort 1 to 4 (Table 1).

Table 1.—Predictor variables and estimates of survival probability (%±SE) to the tailrace of Lower Granite Dam for each cohort of wild subyearling fall chinook salmon, 1998 to 2000. Abbreviations: Date = median day of year of release; Fl = mean fork length (mm) at release; Flow = a flow (m<sup>3</sup>/s) exposure index calculated as the mean flow measured at Lower Granite Dam during the period when the majority of smolts passed the dam; and, Degrees = a water temperature (°C) exposure index calculated as the mean temperature measured in the tailrace of Lower Granite Dam during the period when the majority of smolts passed the dam.

Cohort	N	Date	Fl	Flow	Degrees	Survival
<b>1998</b>						
1	515	140	80	2,344	17.6	70.8±2.9
2	515	141	75	2,021	18.7	66.1±3.3
3	515	153	73	1,898	19.0	52.8±3.1
4	515	167	70	1,299	19.8	35.6±2.9
<b>1999</b>						
1	441	147	80	2,378	16.3	87.7±4.6
2	440	153	77	1,963	17.1	77.0±3.8
3	440	152	70	2,116	16.7	81.2±5.8
4	440	167	68	1,353	18.3	36.4±3.5
<b>2000</b>						
1	303	130	77	1,510	16.7	57.1±4.1
2	302	144	77	1,296	17.6	53.4±4.2
3	302	146	77	1,274	17.8	44.4±3.6
4	302	158	71	859	18.5	35.7±4.3

*Survival Modeling*

Release date and fork length were negatively correlated ( $N = 12$ ;  $r = -0.76$ ;  $P = 0.004$ ). Therefore, release date and fork length were not entered into the same multiple regression model. Fork length and flow ( $N = 12$ ;  $r = 0.47$ ;  $P = 0.12$ ), fork length

and temperature ( $N = 12$ ;  $r = -0.54$ ;  $P = 0.07$ ), and flow and temperature ( $N = 12$ ;  $r = -0.45$ ;  $P = 0.15$ ) were non-collinear.

The model that predicted cohort survival from flow and temperature had a Mallows'  $C_p$  score one less than the number of parameters, the lowest AIC value, and an  $R^2$  of 0.923 (Table 2). The models that included fork length or release date had Mallows'  $C_p$  scores that equaled the number of parameters, relatively low AIC values, and  $R^2$  values of 0.923 (Table 2), but the slope coefficient for fork length ( $t = 0.05$ ;  $P = 0.96$ ) and release date ( $t = 0.07$ ;  $P = 0.94$ ) did not differ significantly from zero.

The final multiple regression model was: Cohort survival =  $140.82753 + 0.02648 \times \text{Flow} - 7.14437 \times \text{Temperature}$ . The final model was significant ( $N = 12$ ;  $P \leq 0.0001$ ) as were the coefficients for flow ( $t = 6.81$ ;  $P \leq 0.0001$ ) and temperature ( $t = -3.96$ ;  $P = 0.003$ ). Flow and temperature explained 92.3% of the observed variability in cohort survival to the tailrace of Lower Granite Dam. Cohort survival generally increased as flow increased, and decreased as temperature increased (Figure 3).

#### *Assessing Summer Flow Augmentation*

Water releases for summer flow augmentation in 1998, 1999, and 2000 were generally timed to passage of later migrating smolts at Lower Granite Dam (Figures 4–6). Therefore, later cohorts were usually predicted to accrue greater survival benefits than earlier cohorts (Table 3). For all cohorts, estimated survival to the tailrace of Lower Granite Dam was predicted to be higher when summer flow augmentation was implemented than when it was not implemented (Table 3; Figure 7).

Table 2.—Mallow's Cp scores, Akaikes information criteria (AIC), and coefficients of determination ( $R^2$ ) used to compare the fit of multiple regression models describing the survival of cohorts of wild subyearling fall chinook salmon from release in the Snake River to the tailrace of Lower Granite Dam, 1998 to 2000. Abbreviations: Date = median day of year of release; Fl = mean fork length (mm) at release; Flow = a flow ( $\text{m}^3/\text{s}$ ) exposure index calculated as the mean flow measured at Lower Granite Dam during the period when the majority of smolts passed the dam; and, Degrees = a water temperature ( $^{\circ}\text{C}$ ) exposure index calculated as the mean temperature measured in the tailrace of Lower Granite Dam during the period when the majority of smolts passed the dam.

C(p)	AIC	$R^2$	Variables in model
2	44	0.923	Flow Degrees
4	46	0.923	Fl Flow Degrees
4	46	0.923	Date Flow Degrees

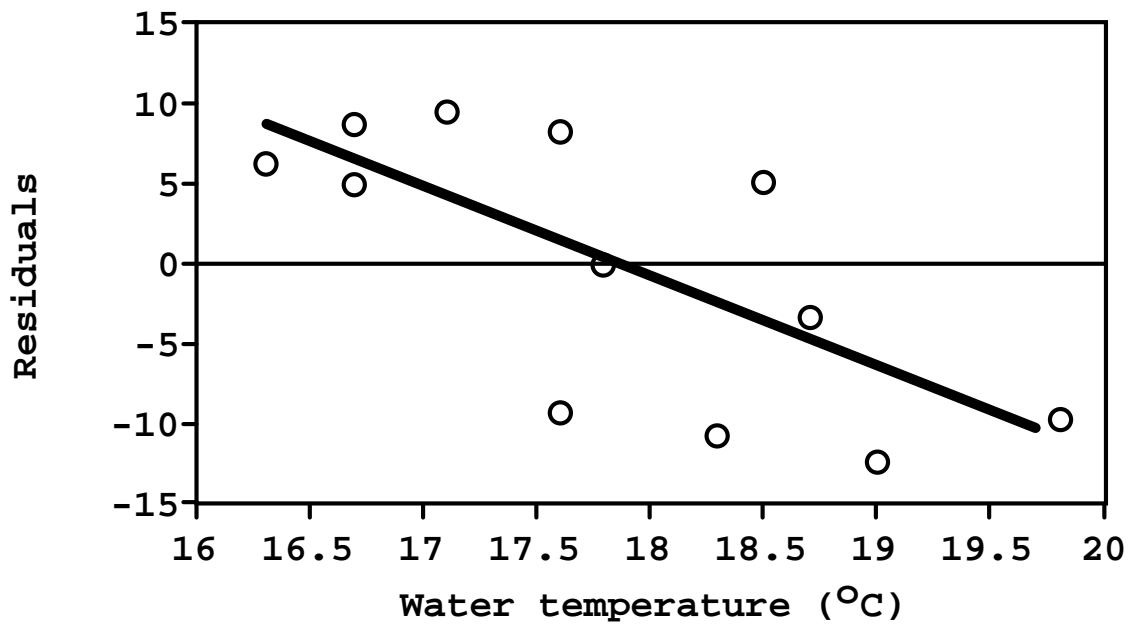
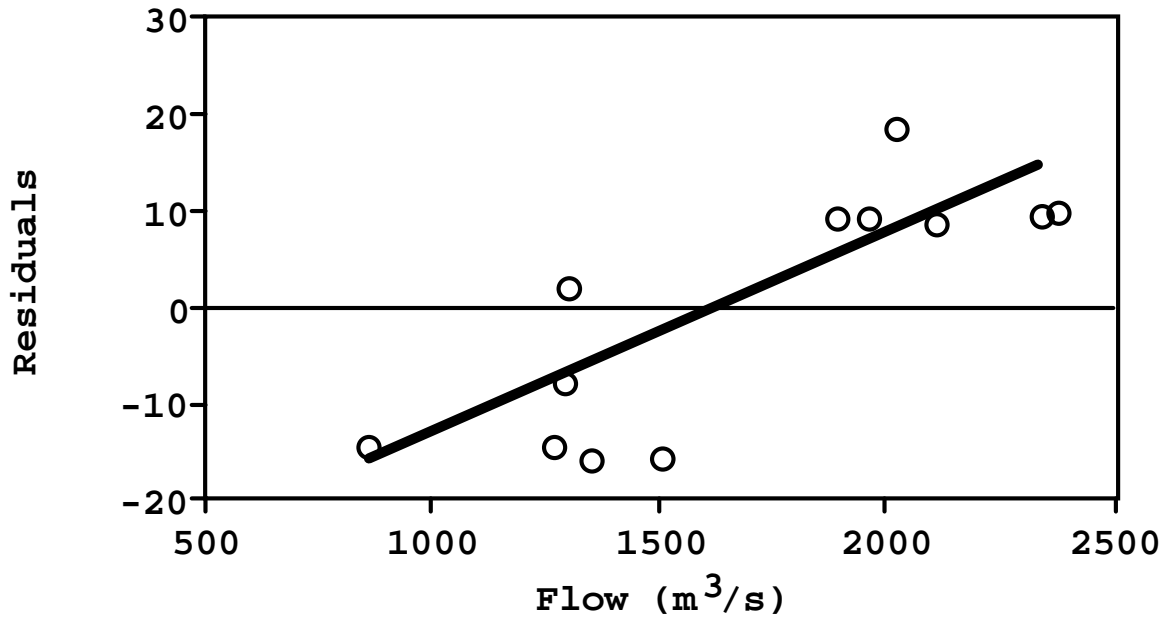


Figure 3.—Residuals plots for flow and water temperature. Residuals are from ordinary least-squares multiple regression models fit to predict cohort survival from the predictor variables that is not on the X axis. The line in each plot was predicted by regressing the residuals against the predictor variable on the X axis.

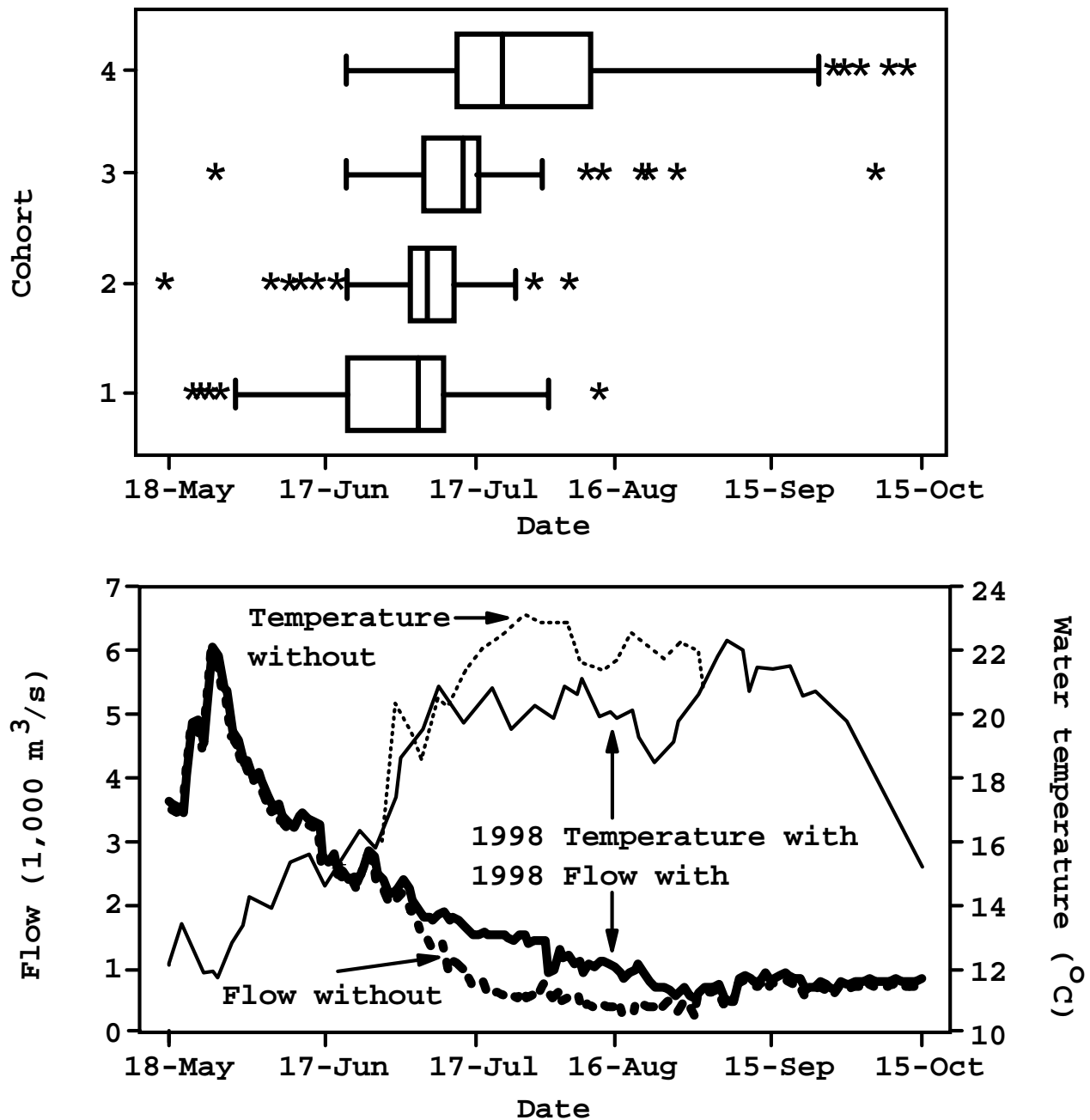


Figure 4.—Box plots showing passage timing at Lower Granite Dam for PIT-tagged wild subyearling fall chinook salmon from each of four cohorts in 1998 (Top), and the mean daily flows and water temperatures observed in Lower Granite Reservoir when flow was augmented (with) compared to those that may have occurred if flows had not been augmented (without; Bottom). See Figure 2 for a description of box plots.



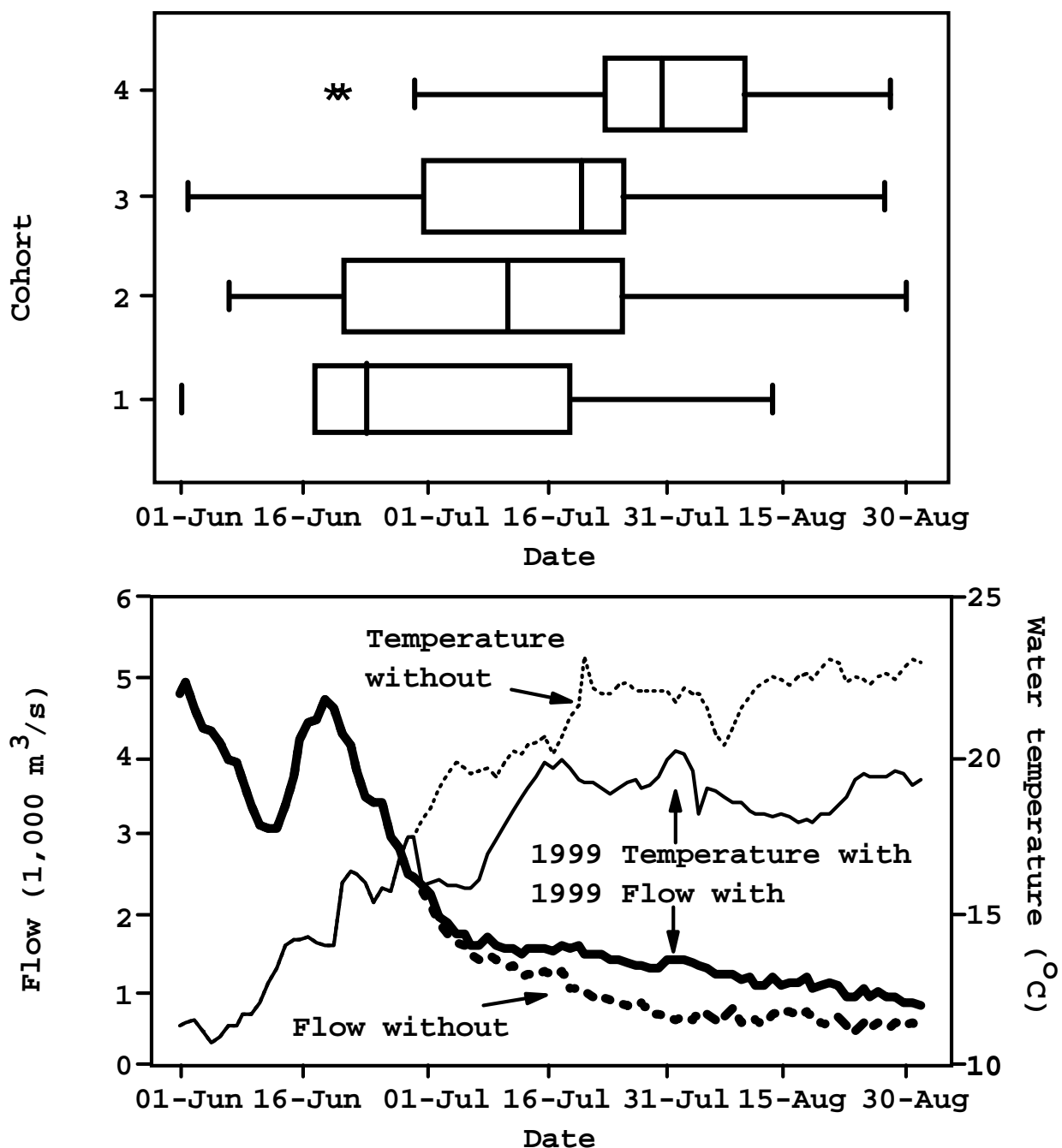


Figure 5.—Box plots showing passage timing at Lower Granite Dam for PIT-tagged wild subyearling fall chinook salmon from each of four cohorts in 1999 (Top), and the mean daily flows and water temperatures observed in Lower Granite Reservoir when flow was augmented (with) compared to those that may have occurred if flows had not been augmented (without; Bottom). See Figure 2 for a description of box plots.

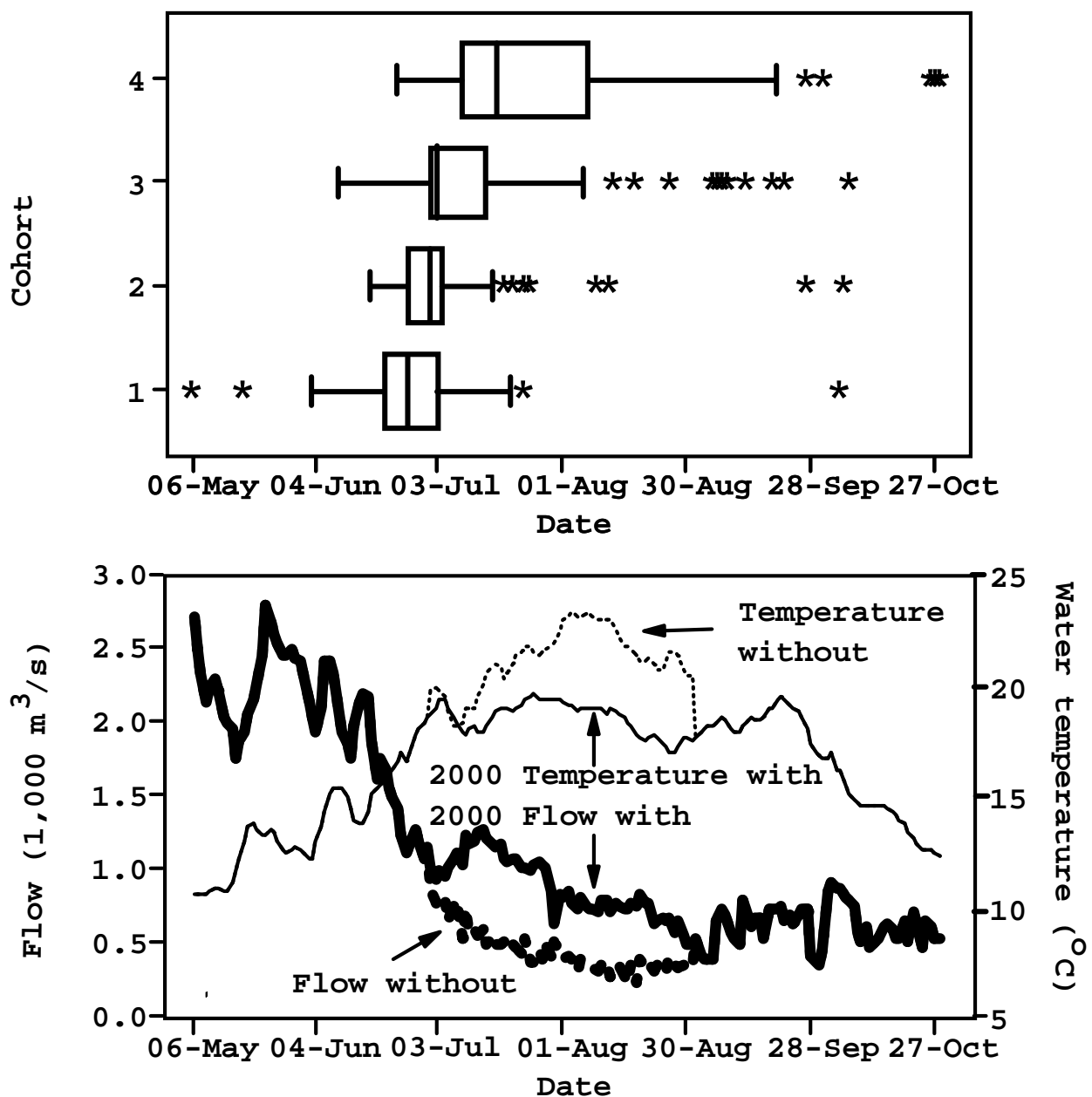


Figure 6.—Box plots showing passage timing at Lower Granite Dam for PIT-tagged wild subyearling fall chinook salmon from each of four cohorts in 2000 (Top), and the mean daily flows and water temperatures observed in Lower Granite Reservoir when flow was augmented (with) compared to those that may have occurred if flows had not been augmented (without; Bottom). See Figure 2 for a description of box plots.

Table 3.—Predicted survival (%+95% C.I.) to the tailrace of Lower Granite Dam for cohorts of wild subyearling fall chinook salmon released in the Snake River from 1995 to 1998. Predictions were made using the observed flow and water temperature indices in Table 1 (Survival with), and by using flow (m<sup>3</sup>/s) and water temperature (°C) exposure indices recalculated to approximate conditions that would have occurred if flow had not been augmented (Survival without).

Cohort	Survival with	Recalculated		Survival without	Difference in survival
		Flow	Temperature		
1998					
1	77.2±6.5	2,066	18.3	64.8±5.8	12.4
2	60.7±6.6	1,689	19.3	47.7±7.0	13.0
3	55.3±6.8	1,468	20.1	36.1±9.3	19.2
4	33.8±8.0	988	21.3	14.8±13.1	19.0
1999					
1	87.3±7.5	2,128	17.1	75.0±5.2	12.3
2	70.6±4.7	1,667	18.4	53.5±4.3	17.1
3	77.5±5.8	1,837	18.0	60.9±4.0	16.6
4	45.9±4.6	943	20.1	22.2±9.4	23.7
2000					
1	61.5±6.7	1,314	17.0	54.2±6.8	7.3
2	49.4±5.5	1,078	17.9	41.5±6.5	7.9
3	47.4±5.3	978	18.6	33.8±6.7	13.6
4	31.4±7.5	587	20.1	12.8±10.6	18.6

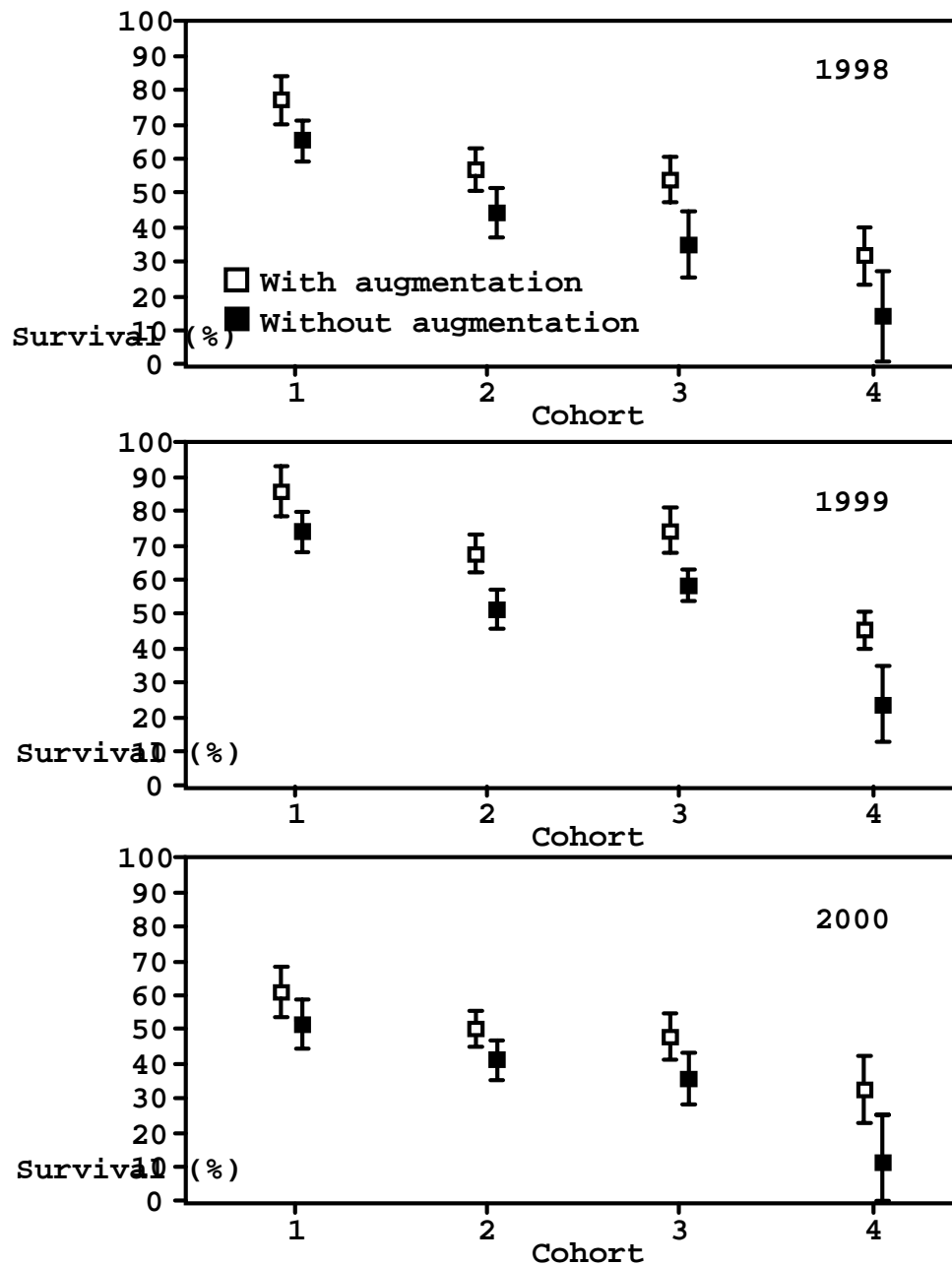


Figure 7.—Survival ( $\pm 95\%$  C.I.) to the tailrace of Lower Granite Dam for PIT-tagged wild subyearling fall chinook salmon (1998 Top; 1999 Middle; 2000 Bottom) predicted from observed mean flow and water temperatures (from Table 1), and from mean flows and water temperatures recalculated to represent those that would have occurred if flow were not augmented (from Table 3). The equation Cohort survival =  $140.82753 + 0.02648 \times \text{Flow} - 7.14437 \times \text{Temperature}$  was used to make both sets of predictions.

## Discussion

Survival of wild subyearling fall chinook salmon from release in the Snake River to the tailrace of Lower Granite Dam generally increased as flow increased and decreased as temperature increased. Based on the regression model we developed, survival is predicted to change by approximately 3% with each change of 100 m<sup>3</sup>/s in flow when temperature is held constant. The change in survival is approximately 7% for each 1°C increase or decrease in temperature when flow is held constant. Kjelson et al. (1982), Kjelson and Brandes (1989), and Connor et al. (1998) also reported that survival of subyearling chinook salmon during seaward migration is directly proportional to flow and inversely proportional to temperature.

Flow and temperature were closely correlated in the above three studies (e.g., Connor et al. 1998;  $r = -0.999$ ), thus the researchers could not determine if the high correlation between survival and one variable was caused by the other variable. Flows and temperatures were atypically non-collinear ( $r = -0.45$ ) during 1998 to 2000, therefore we were able to enter both of these predictor variables in the same multiple regression equation without biasing the regression coefficients. Both regression coefficients differed significantly from zero (flow  $P \leq 0.0001$ ; temperature  $P = 0.003$ ). We conclude that flow and temperature act together to influence fall chinook salmon survival.

Correlation does not imply causation unless the causal mechanisms can be identified with certainty. Flow and water temperature, however, are the two most plausible factors affecting survival since fall chinook salmon are aquatic poikilotherms. We suggest that the two variables assert their influence on survival simultaneously. For example, flow influences downstream migration rate (Berggren and Filardo 1993; Chapter Five in this report) and water turbidity at the same time temperature is regulating predation (Vigg and Burley 1991; Curet 1994; Anglea 1997). Fall chinook salmon that migrate downstream when flow is low and temperatures are warm might suffer high mortality because they are exposed for longer durations to actively feeding predators in clear water.

Slow downstream movement and late-summer passage associated with low flow levels (Chapter Five in this report) can also result in exposure to temperatures above 20°C. Prolonged exposure to temperatures above 20°C might disrupt fall chinook salmon growth, smoltification, and downstream movement, thereby exacerbating predation (Marine 1997). Temperatures above 20°C have also been associated with disease and stress-induced mortality (W. P. Connor, unpublished data).

## Management Implications

Discussing the management implications of the results in this paper requires an understanding of the limitations on our study. We could not ascertain where PIT-tagged fall chinook salmon died en route to Lower Granite Dam. It was also impossible to determine where tagged fish spent the majority of time between release and detection at the dam. Our assessment of summer flow augmentation would be weakened if the majority of tagged fish died in the free-flowing Snake River before flow was augmented. On the other hand, the effect of summer flow augmentation on survival may have been underestimated because observed passage dates were used when recalculating flow and water temperature exposure indices. Estimates suggest that smolts passed Lower Granite Dam up to 49 d earlier when summer flow augmentation was implemented, than when it was not implemented (Chapter Five in this report). Therefore, the recalculated flow exposure indices used in this paper were probably too high, the water temperature exposure indices were too low, and survival predictions made using these indices were probably higher than would be the case if flows had not been augmented.

In spite of the above limitations, we believe the results in this paper support summer flow augmentation as a beneficial interim recovery measure for Snake River fall chinook salmon. Survival for all 12 cohorts was predicted to be higher when flow augmentation was implemented than when flow was not augmented. We conclude that increases in flow and decreases in water temperature resulting from summer flow augmentation increase fall chinook salmon smolt survival.

Although summer flow augmentation likely increased survival of fall chinook salmon passing downstream in Lower Granite Reservoir, smolt mortality is likely still higher than before dams were constructed. When the lower Snake River was still free-flowing chinook salmon smolts were exposed to minimum flows of approximately 3,400 and 5,000 m<sup>3</sup>/s in 1954 and 1955, respectively (Mains and Smith 1964). Maximum temperatures for 1954 and 1955 were approximately 16 and 13°C (Mains and Smith 1964). In contrast, the maximum flow and minimum temperature experienced by fall chinook salmon cohorts from 1998 to 2000 were 2,510 m<sup>3</sup>/s and 16.3°C.

Releasing larger volumes of cooler reservoir water during the summer would provide present-day fall chinook salmon smolts with migration conditions more similar to their pre-dam counterparts. Dworshak Reservoir, and reservoirs upstream of Brownlee Dam, however, are the only two sources of additional water. The ability of fishery managers to obtain more cool water for summer flow augmentation from Dworshak Reservoir is limited by technical, political, and biological constraints. Dworshak Reservoir is routinely drafted to near minimum

operation levels, so releasing more water would reduce the probability of refill the next year. Releasing larger volumes of water from Dworshak Reservoir earlier in the year to cover a larger percentage of the smolt migration would be difficult because political opponents of summer flow augmentation advocate maintaining the reservoir at full-pool as long after the 4<sup>th</sup> of July as possible. Releasing colder water from Dworshak Reservoir using its multi-level selector gates would likely disrupt growth and seaward movement of fall chinook salmon that are still rearing in the lower Clearwater River when smolts from the Snake River are passing downstream in Lower Granite Reservoir (Chapter Four in this report). For example, the release of 6°C water in July 1994 decreased temperature in Lower Granite Reservoir from approximately 23 to 17°C (Connor et al. 1998), thereby improving conditions for survival of smolts from the Snake River. However, the 6°C release also caused water temperature in the lower Clearwater River to decrease from approximately 19 to 8°C (U. S. Geological Survey data collected at Spalding, Idaho) at a time when young fall chinook salmon were still rearing along the shoreline.

Increasing the supply of water available from reservoirs upstream of Brownlee Dam for summer flow augmentation would be difficult because political opponents of summer flow augmentation advocate using reservoir water for irrigation, power production, and recreation. Cooler water cannot be released from Brownlee Reservoir because Brownlee Dam does not have multi-level selector gates. Consequently, the water released from Brownlee Reservoir for summer flow augmentation is relatively warm (e.g., 17.5 to 20.3°C; Connor et al. 1998). Developing the ability to selectively release cooler water from Brownlee Reservoir might be the most practical option for improving the effectiveness of summer flow augmentation provided that cooler water is available and impacts on native resident fishes would be minimal. Cool water could be released from Brownlee Reservoir when fall chinook salmon smolts from the Snake River are passing downstream in Lower Granite Reservoir without affecting water temperatures in the lower Clearwater River when fry and parr are still rearing.

### **Acknowledgments**

D. Steele, C. Eaton, R. Nelle, R. Waitt, C. Larsen, J. Bretz, S. Bradbury, C. Piston, and many other employees of the U. S. Fish and Wildlife Service's Idaho Fishery Resource Office collected data. The staff at the Fish Passage Center of the Columbia River Basin Fish and Wildlife Authority provided flow data. C. Stein, D. Marvin, and the staff at the Pacific States Marine Fisheries Commission managed the PIT-tag data base. D. Bennett, P. Bigelow, and J. Congleton reviewed drafts of the manuscript. The editor, an associate editor, and three anonymous reviewers improved the manuscript. Funding was the rate payers of the Bonneville Power Administration through Contract Number DE-AI79-91BP21708 administered by D. Docherty, M. Galloway, D. Praest, M. Beeman, and K. Tiffan.



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Appendix 1.—Mean daily flows (m<sup>3</sup>/s) in Lower Granite Reservoir with and without summer flow augmentation, 1998 to 2000.

Date	1998		1999		2000	
	With	Without	With	Without	With	Without
01-Jul	2195	2138	2336	2243	1020	892
02-Jul	2212	2127	2212	2050	952	790
03-Jul	2251	2130	1931	1863	1014	835
04-Jul	2419	2283	1832	1702	977	816
05-Jul	2274	2116	1699	1594	1020	677
06-Jul	2065	1957	1685	1546	1090	773
07-Jul	1960	1844	1563	1427	1121	793
08-Jul	1827	1592	1546	1385	1059	552
09-Jul	1801	1515	1648	1458	1246	753
10-Jul	1778	1436	1563	1357	1198	583
11-Jul	1866	1385	1509	1269	1204	612
12-Jul	1892	1504	1532	1294	1274	572
13-Jul	1745	1087	1447	1136	1280	600
14-Jul	1812	1198	1529	1184	1229	513
15-Jul	1759	1164	1507	1172	1184	561
16-Jul	1651	1073	1507	1212	1161	501
17-Jul	1583	9711	475	1136	1187	507
18-Jul	1555	8301	541	1238	1087	524
19-Jul	1549	8441	501	991	1073	470
20-Jul	1577	8811	546	988	1099	504
21-Jul	1521	7391	456	954	1096	490
22-Jul	1535	7191	453	912	1028	450
23-Jul	1549	7141	456	895	1028	541
24-Jul	1512	6881	376	847	1005	382
25-Jul	1481	6851	354	824	1051	399
26-Jul	1444	6461	345	787	1076	467
27-Jul	1521	6571	314	762	1042	416
28-Jul	1529	7621	308	824	1031	515
29-Jul	1410	6151	257	685	860	436
30-Jul	1453	6661	263	671	643	530

Appendix 1.-(Continued)

31-Jul	1439	649	1368	634	855	453
01-Aug	1450	830	1357	617	833	408
02-Aug	954	765	1382	632	864	428
03-Aug	963	612	1323	615	784	402
04-Aug	1283	705	1303	702	748	337
05-Aug	1167	586	1266	660	833	413
06-Aug	1201	634	1175	615	776	360
07-Aug	1065	592	1181	640	759	351
08-Aug	1107	671	1198	753	745	354
09-Aug	943	436	1116	555	733	326
10-Aug	1065	510	1141	671	813	362
11-Aug	1045	484	1054	600	813	377
12-Aug	1104	524	1028	547	733	280
13-Aug	1136	552	1164	694	787	368
14-Aug	1087	496	1028	697	773	362
15-Aug	1028	496	1090	702	750	297
16-Aug	960	524	1073	657	753	261
17-Aug	827	396	1170	711	799	365
18-Aug	954	445	1022	595	767	252
19-Aug	974	413	1025	578	858	408
20-Aug	1065	566	1070	544	787	354
21-Aug	932	521	1051	637	787	391
22-Aug	787	487	906	538	649	329
23-Aug	716	498	898	462	677	365
24-Aug	719	490	997	569	691	354
25-Aug	688	487	892	487	671	331
26-Aug	683	552	960	569	685	428
27-Aug	575	462	901	467	583	360
28-Aug	617	402	912	583	677	354
29-Aug	697	544	827	527	566	362
30-Aug	592	541	810	552	513	346
31-Aug	507	334	782	476	518	368

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Appendix 2.—Mean water temperatures (°C) in Lower Granite Reservoir with and without summer flow augmentation, 1998 to 2000.

Date	1998		1999		2000	
	With	Without	With	Without	With	Without
01-Jul 16.6	19.0		15.8	16.2	18.8	17.8
02-Jul 17.5	19.8		15.9	16.6	19.1	18.2
03-Jul 18.1	20.1		16.0	16.9	19.4	18.7
04-Jul 18.7	20.1		15.8	16.8	19.4	18.9
05-Jul 19.0	20.3		15.8	17.0	19.0	19.2
06-Jul 19.0	20.1		15.7	17.0	18.7	19.3
07-Jul 19.3	19.7		15.7	16.8	18.4	20.0
08-Jul 19.7	19.7		16.0	17.0	18.0	20.1
09-Jul 20.1	19.5		16.8	16.7	17.9	20.3
10-Jul 20.6	19.7		17.3	17.1	18.1	19.7
11-Jul 20.7	19.5		17.7	17.3	18.3	19.2
12-Jul 20.8	20.0		18.2	18.1	18.0	19.3
13-Jul 20.5	20.4		18.6	18.5	18.0	19.3
14-Jul 20.2	20.6		18.9	18.7	18.2	19.1
15-Jul 20.0	20.7		19.3	19.0	18.6	19.0
16-Jul 19.7	20.7		19.7	19.3	18.9	18.8
17-Jul 19.9	20.7		19.6	19.8	19.1	19.3
18-Jul 19.9	20.8		19.8	20.1	19.0	19.6
19-Jul 20.4	20.9		19.6	20.3	19.0	19.7
20-Jul 20.4	21.3		19.2	20.2	18.9	19.9
21-Jul 20.9	21.8		19.1	19.9	19.1	20.3
22-Jul 20.7	22.0		19.1	19.9	19.2	20.3
23-Jul 20.1	22.2		18.9	19.7	19.4	20.2
24-Jul 19.7	22.4		18.7	19.8	19.6	20.6
25-Jul 19.5	22.6		18.9	19.5	19.7	20.8
26-Jul 19.7	22.7		19.1	19.3	19.5	21.0
27-Jul 19.7	23.0		19.2	19.4	19.4	21.2
28-Jul 19.7	22.9		18.9	19.9	19.5	21.2
29-Jul 20.2	23.1		19.0	21.0	19.5	21.6
30-Jul 20.1	23.3		19.3	21.2	19.4	21.7

Appendix 2.-(Continued)

31-Jul	20.2	23.7	19.8	20.8	19.4	21.8
01-Aug	20.0	23.8	20.1	21.0	19.3	22.0
02-Aug	19.9	23.9	20.0	21.2	19.2	21.9
03-Aug	20.0	24.0	19.5	21.2	19.2	22.0
04-Aug	20.2	24.3	18.1	21.3	18.9	22.3
05-Aug	21.0	24.4	18.9	21.2	19.0	22.6
06-Aug	20.9	24.1	18.8	21.8	19.1	22.4
07-Aug	20.7	23.9	18.6	22.4	19.0	22.6
08-Aug	21.0	23.5	18.5	22.6	19.0	22.8
09-Aug	21.2	23.5	18.5	22.6	19.0	22.5
10-Aug	20.8	23.4	18.2	23.2	19.0	22.5
11-Aug	20.1	23.2	18.1	22.8	18.8	22.6
12-Aug	19.9	23.3	18.1	22.9	19.0	22.4
13-Aug	20.0	23.3	18.0	22.8	18.9	22.6
14-Aug	20.2	23.4	18.1	22.8	18.8	23.0
15-Aug	20.0	23.6	18.0	22.7	18.6	23.1
16-Aug	19.9	23.4	17.8	22.3	18.4	23.2
17-Aug	20.0	23.1	17.9	22.2	18.3	23.4
18-Aug	19.9	22.6	17.8	22.1	17.8	23.3
19-Aug	19.8	22.3	18.1	21.9	17.7	23.2
20-Aug	19.3	22.2	18.1	21.9	17.6	23.0
21-Aug	18.9	22.4	18.4	21.9	17.7	23.0
22-Aug	18.7	22.4	18.6	22.1	17.8	23.0
23-Aug	18.5	22.5	19.2	21.5	17.7	22.6
24-Aug	18.6	22.3	19.4	21.1	17.5	22.9
25-Aug	18.6	22.0	19.3	20.9	17.4	22.7
26-Aug	18.8	22.2	19.3	20.9	17.1	22.5
27-Aug	18.9	21.8	19.3	20.6	17.0	22.2
28-Aug	19.5	21.9	19.5	20.6	17.4	22.0
29-Aug	19.9	21.5	19.4	21.4	17.7	22.0
30-Aug	20.0	21.7	19.0	21.9	17.7	21.7
31-Aug	20.4	21.5	19.2	21.9	17.6	21.5

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## CHAPTER SEVEN

Growth of Wild Subyearling Fall Chinook Salmon  
in the Snake River\*

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\*Submitted to the North American Journal of Fisheries Management



*Abstract.*—Growth is an important determinant of life history development for juvenile anadromous salmonids. Dams can affect growth by displacing spawners and juveniles into less productive habitat. In the Snake River, dams extirpated fall chinook salmon *Oncorhynchus tshawytscha* spawners from the most productive area into habitat with lower growth opportunity. We collected fall chinook salmon juveniles in two reaches of the Snake River to describe growth in fork length (mm/d), and to test for a relation between growth and water temperature. Growth rate during shoreline rearing was significantly ( $P = 0.003$ ) higher for parr in the warmer of these two reaches (grand means  $1.2 \pm 0.04$  and  $1.0 \pm 0.04$  mm/d). Smolts from the two reaches share a common relatively warm downstream migration route, thus growth rates were similar between smolts from the two reaches ( $P = 0.18$ ; grand means  $1.3 \pm 0.04$  and  $1.4 \pm 0.04$  mm/d). By pooling data across reaches and life stages, we found that growth rate generally increased as water temperature increased ( $N = 17$ ;  $r^2 = 0.62$ ;  $P = 0.0002$ ). The growth rates we observed were probably lower than for fall chinook salmon in the historical rearing area, but they were still rapid by comparison to those reported for ocean-type chinook salmon in presumably more productive brackish and saltwater habitats. We suggest that growth could be used to index the well-being of Snake River fall chinook salmon.

## Introduction

Construction and operation of dams has affected many stocks of anadromous salmonids in the North America (e.g., Moffitt et al. 1982; Wunderlich et al. 1994; Kondolf et al. 1996; Dauble and Watson 1997). Numerous dams were constructed along the upper Snake River in southern Idaho throughout the 20<sup>th</sup> century. Completion of seven dams from 1958 to 1975 markedly reduced the potential of the Snake River for fall chinook salmon *Oncorhynchus tshawytscha*. Brownlee, Oxbow, and Hells Canyon dams eliminated spawning and rearing in the most productive habitat upstream of Marsing, Idaho (Groves and Chandler 1999; Dauble and Geist 2000; Figure 1). Lower Granite, Little Goose, Lower Monumental, and Ice Harbor dams further reduced spawning and rearing habitat availability by impounding the lower 224 km of the Snake River (Figure 1).

The majority of wild subyearlings that inhabit the 173 km of riverine habitat between Hells Canyon Dam and the upper end of Lower Granite Reservoir (Figure 1) are offspring of fall chinook salmon spawners (Marshall et al. 2000; Connor et al. 2001a; W. P. Connor, unpublished data). The remaining subyearlings are wild spring and summer chinook salmon that disperse long distances from natal streams into the Snake River where they rear, grow rapidly, and then migrate seaward a little earlier than fall chinook salmon (Connor et al. 2001a, 2001b). For simplicity, we refer to the wild subyearling chinook salmon that inhabit the shorelines of the Snake River during spring and summer as fall chinook salmon.

In Chapter Four of this report, we described how the upper reach (Figure 1) of the Snake River was warmer than the lower reach during winter through spring when eggs were incubating, and during spring when juveniles were rearing and starting seaward movement. Consequently, life history of young fall chinook salmon progressed on an earlier time schedule in the upper reach of the Snake River than in the lower reach of the Snake River. Assuming life stage progression was a crude measure of growth, we concluded in Chapter Four of this report that relatively warmer water temperatures and higher growth opportunity (Metcalf and Thorpe 1990; Taylor 1990) in the upper reach of the Snake River explained the differences in life stage development we observed.

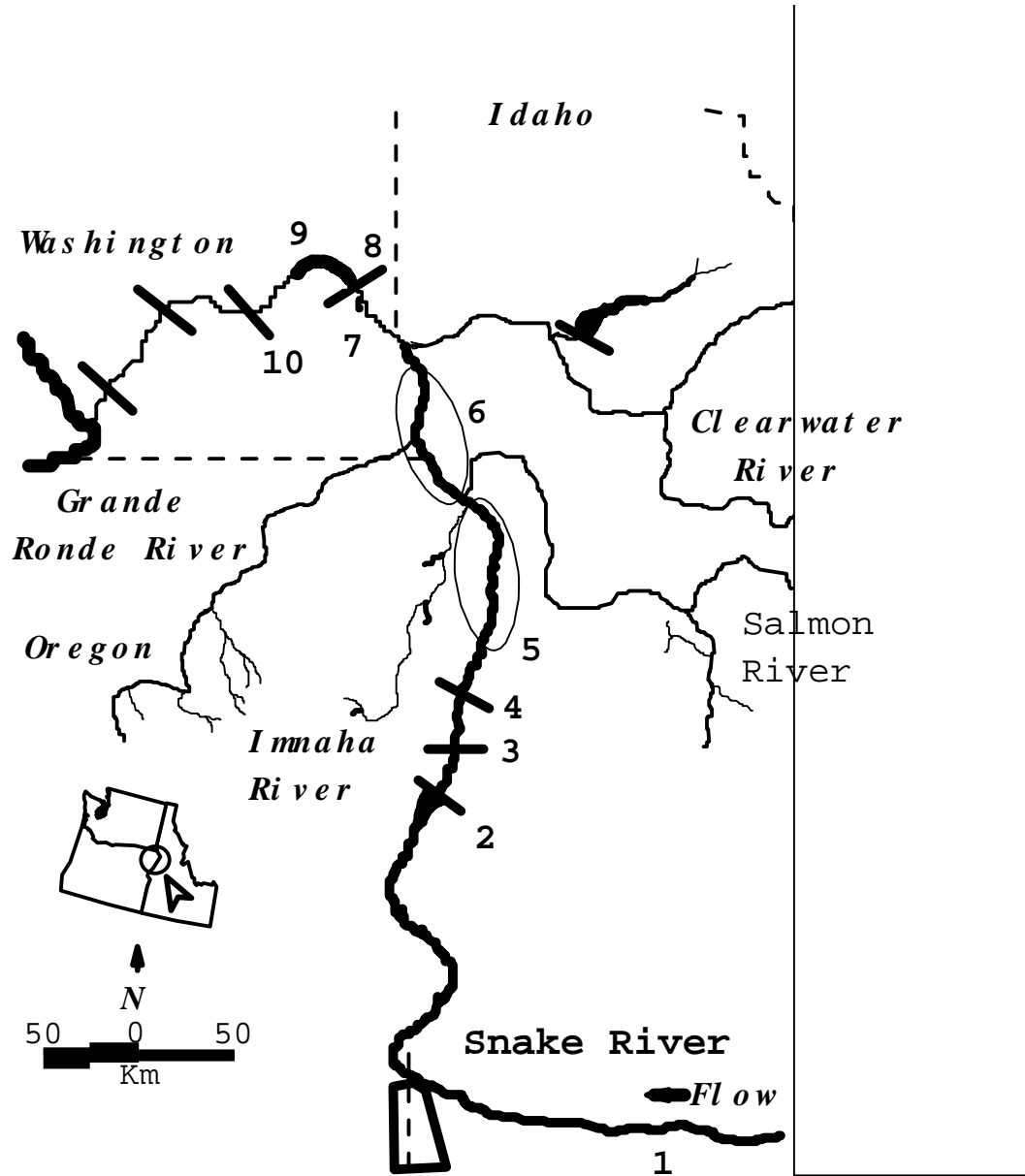


Figure 1.—Locations of the upper and lower reaches of the Snake River where adult fall chinook salmon spawn and their offspring were captured by using a beach seine (cross hatched ellipses), and Lower Granite and Little Goose dams and reservoirs. The locations are as follows: 1 = historical spawning area near Marsing, Idaho; 2) Brownlee Dam; 3) Oxbow Dam; 4) Hells Canyon Dam; 5) Snake River upper reach; 6) Snake River lower reach; 7) Lower Granite Reservoir; 8) Lower Granite Dam; 9) Little Goose Reservoir; and 10) Little Goose Dam.

In this chapter, we expand on the findings in Chapter Four of this report by estimating and comparing growth in fork length for fall chinook salmon in the upper and lower reaches of the Snake River. We also test for a relation between water temperature and growth in fork length.

### **Methods**

Water temperature.—Data were collected in the Snake River from 1995 to 1999 using hourly recording thermographs stationed known distances (rkm) upstream from each river mouth. Thermographs were typically stationed offshore in relatively deep water to ensure submergence at all flow levels. Thermograph locations in the Snake River varied by year and flow level. Data were collected at rkm 383, rkm 369, rkm 325, and rkm 303 in the upper reach of the Snake River, and at rkm 290, rkm 287, rkm 274, rkm 265, and rkm 251 in the lower reach of the Snake River. No temperature data were available for the lower reach of the Snake River in 1996 or either reach in 2000 due to thermograph failure.

Daily mean water temperature was calculated from thermograph output. Data for two or more thermographs in the Snake River were averaged within a reach to provide one daily mean water temperature value. Missing daily mean values were predicted by using ordinary least-squares regression (range of  $r^2$  values 0.94 to 0.99). For example, missing daily mean values were predicted for 6/30/99 to 7/7/99 based on a regression model fit using observed day of year (e.g., January 1 = 1) and daily mean water temperatures collected three weeks before 6/30/99 and three weeks after 7/7/99.

Daily mean water temperature data were also collected by the U. S. Army Corps of Engineers in the tailrace of Lower Granite Dam from 1995 to 1998. We used data collected in the forebay of Lower Granite Dam when tailrace data were unavailable.

Two water temperature indices were calculated from the daily mean water temperature data. Mean spring (3/20 to 6/20) water temperatures in each reach of the Snake River were used to index growth conditions during shoreline rearing. Mean spring-summer (3/20 to 9/21) water temperatures at Lower Granite Dam were used to index growth conditions during seaward migration.

Growth.—Fall chinook salmon were captured in the upper and lower reaches of the Snake River from 1992 to 2000. We analyzed

data collected on fall chinook salmon from 1995 to 2000 because data sets were complete for both the upper and lower reaches of the Snake River. Sampling was conducted using a beach seine (Connor et al. 1998). Beach seining typically started in April soon after fry began emerging from the gravel, and was conducted 1 d/week at permanent stations within each spawning and rearing area. Once a majority of fish were at least 60 mm fork length, additional stations were sampled in each spawning and rearing area for three consecutive weeks. Sampling was discontinued in June or July when the majority of fish had moved into Lower Granite Reservoir or to points further downstream.

Passive integrated transponders (PIT) tags (Prentice et al. 1990b) were inserted into fall chinook salmon parr 60 mm fork length and longer (Connor et al. 1998). Tagged parr were released at the collection site after a 15-min recovery period. Some of these PIT-tagged parr were recaptured at beach seining stations up to 46 d after initial capture. We calculated growth in fork length(mm/d) for parr as: fork length at recapture minus fork length at initial capture divided by the number of days between initial capture and recapture.

Some of the PIT-tagged fish were detected as smolts as they passed downstream in the juvenile bypass systems of dams equipped with PIT-tag monitors (Matthews et al. 1977; Prentice et al. 1990a; Figure 1). We recaptured a subsample of the detected smolts using a diversion device (Downing et al. 2001) located in the fish bypass system of Lower Granite Dam in 1995, and Little Goose Dam from 1996 to 1998. We calculated growth rate for smolts using the equation described for parr.

*Statistical analyses.*—We calculated grand mean growth rates by reach and life stage event. For example, grand mean growth rate for parr in the upper reach of the Snake River was calculated as the mean of the six mean annual growth rates for parr in the upper reach. Grand mean growth rate for the parr life stage was calculated as the mean of the 12 mean annual growth rates for parr of both reaches.

We used ANOVA ( $\alpha = 0.05$ ) with a randomized block design blocking on year to test three null hypotheses: 1) there is no difference in growth rates of parr in the upper and lower reaches of the Snake River; 2) there is no difference in growth rates of smolts originating from the upper and lower reaches of the Snake River; 3) there is no difference in growth rates of parr and smolts. Tukey-type pair-wise comparisons ( $\alpha =$

0.05) were made to test for significant differences between grand means.

We used ordinary least-squares regression ( $\alpha = 0.05$ ) to test the relation between growth rate and water temperature. Mean growth rates were pooled across reaches and life stage events to provide the dependent variable in this regression. Spring water temperature was the independent variable paired parr growth rates. Spring-summer water temperature was the independent variable paired with smolt growth rates.

## Results

Water temperatures during spring were warmer in the upper reach of the Snake River than in the lower reach (Table 1). Water temperatures measured at Lower Granite Dam during the spring-summer period were warmer than those measured in the both reaches of the Snake River during spring (Table 1).

During the 6 years, PIT tags were inserted into 7,506 fall chinook salmon parr. Of these, 1,028 were recaptured (Table 2). Approximately 80% of the parr used to calculate growth rates were tagged and then recaptured during the spring. Fork length of PIT-tagged parr during shoreline rearing averaged 69 mm (range 60 to 105 mm). Growth rate was significantly ( $P = 0.003$ ) higher for parr in the upper reach than for parr in the lower reach (Table 2).

A total of 531 PIT-tagged smolts was recaptured at Lower Granite and Little Goose dams combined (Table 3). Approximately 99% of these recaptured smolts passed the dams during the spring and summer. Fork length of recaptured smolts averaged 141 mm (range 84 to 213 mm). The reach of the Snake River where smolts were initially captured and PIT tagged had no effect on growth rate ( $P = 0.18$ ; Table 3).

Grand mean growth rates were  $1.1 \pm 0.05$  mm/d and  $1.3 \pm 0.03$  mm/d for parr and smolts, respectively. Parr growth rates were significantly lower ( $P = 0.002$ ) than smolt growth rates.

Growth rate was significantly related to water temperature ( $P = 0.0002$ ). Water temperature explained 62% of the variability observed in growth rate (Figure 2). Growth rate generally increased as water temperature increased (Figure 2).

Table 1.—Mean spring water temperatures (°C) measured in the upper and lower reaches of the Snake River, and mean spring-summer water temperatures measured at Lower Granite Dam, 1995 to 1999. Grand mean water temperatures  $\pm$  SE are also given.

Year	Upper reach Snake River	Lower reach Snake River	Lower Granite Reservoir
1995	11.8	10.9	15.0
1996	12.7		15.3
1997	12.4	11.2	14.4
1998	12.0	11.5	15.5
1999	12.3	10.6	
Grand means	12. $\pm$ 0.2	11.1 $\pm$ 0.2	15.1 $\pm$ 0.2

Table 2.—Growth rates (mm/d) for wild fall chinook salmon parr in the upper and lower reaches of the Snake River, 1995 to 2000. Data are given as sample size; mean; range. The grand mean  $\pm$  S.E. growth rates were significantly different ( $P = 0.003$ ).

Year	Upper reach	Lower reach
1995	148; 1.2; 0.1-1.8	78; 1.0; 0.0-1.8
1996	19; 1.1; 0.6-1.5	49; 0.9; 0.2-1.9
1997	20; 1.3; 0.9-1.7	80; 0.8; 0.2-1.4
1998	112; 1.1; 0.2-1.6	129; 0.9; 0.0-1.6
1999	171; 1.3; 0.3-1.9	92; 1.1; 0.1-1.7
2000	90; 1.3; 0.8-1.6	40; 1.0; 0.3-1.5
Grand means	1.2 $\pm$ 0.04	1.0 $\pm$ 0.04

Table 3.—Growth rates (mm/d) for wild fall chinook salmon smolts originating from the upper and lower reaches of the Snake River, 1995 to 1998. Data are given as sample size; mean; range. The grand mean  $\pm$  S.E. growth rates were not significantly different ( $P = 0.18$ ).

Year	Upper reach	Lower reach
1995	132; 1.3; 0.7-1.7	156; 1.4; 0.8-2.1
1996	9; 1.3; 1.1-1.5	48; 1.3; 0.7-1.8
1997	19; 1.2; 1.0-1.5	62; 1.3; 0.5-1.7
1998	105; 1.4; 0.8-1.8	146; 1.4; 0.3-2.3
Grand means	1.3 $\pm$ 0.04	1.4 $\pm$ 0.04



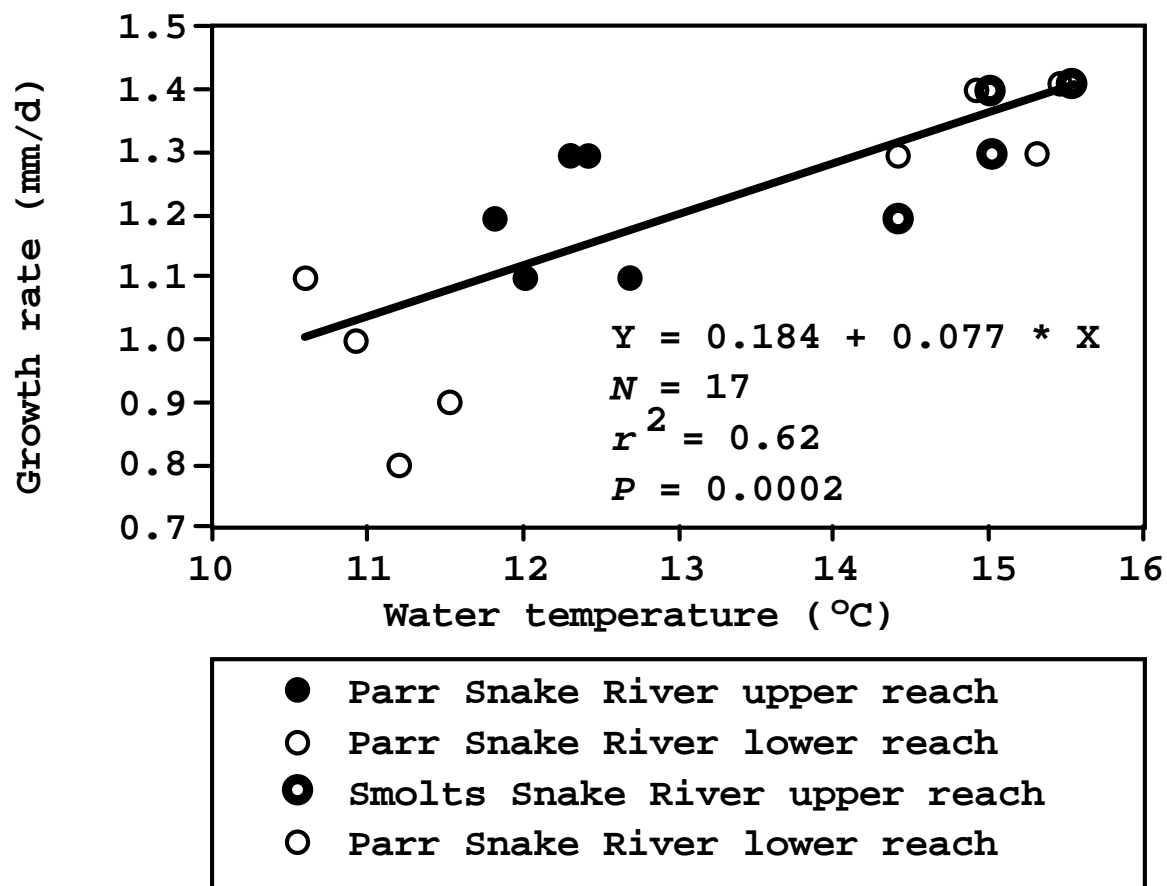


Figure 2.—The relation between juvenile fall chinook salmon growth rate and water temperature. Data for the regression are given in Tables 1, 2, and 3.

## Discussion

Our study was subject to several limitations. Sample sizes of parr and smolts used to calculate growth rates were small in some cases due to low abundance, difficult sampling conditions, and logistical constraints imposed by the dams. We did not analyze all the factors that affect growth of juvenile anadromous salmonids. Water temperature was a logical variable to study because young fall chinook salmon are aquatic poikilotherms, thus their growth is regulated by water temperature (Banks et al. 1971; Marine 1997). We used thermograph data collected offshore in the Snake River, and at Lower Granite Dam to index water temperature conditions experienced by young fall chinook salmon. The actual water temperatures experienced by parr during rearing and smolts during seaward migration were not completely represented by these indices. Local water temperature data collected where fall chinook salmon rear and pass downstream would have improved our regression analysis, but such data would be difficult to collect and were not available from 1995 to 2000.

In spite of the above limitations, we found that differences in fall chinook salmon growth observed between reaches and life stage events could be partly explained by water temperature as proposed in Chapter Four of this report. The upper reach of the Snake River fostered higher rates of growth for parr than the lower reach partly because it was warmer. Parr growth was slower than smolt growth partly because water temperatures during shoreline rearing were cooler than during seaward migration. Smolts originating from the upper and lower reaches of the Snake River grew at approximately the same rates partly because they shared a common relatively warm migration environment.

The level of growth sustained by young anadromous salmonids plays a role on life history development. Some researchers maintain that age at seaward migration decreases as water temperature during rearing increases because growth increases (Metcalf and Thorpe; Taylor 1990; Connor et al. 2001b). In Chapter Four of this report, we found that on average 11% of the fall chinook salmon that were PIT tagged in the lower reach of the Snake River did not complete seaward migration until they were yearlings, compared to only 2% for fish in the upper reach. This suggests a temperature-related tendency toward a stream-type (Healey 1991) life history that could reduce survival by extending freshwater residency. Differences in parr growth

rates caused by water temperature provide a plausible explanation for this observation.

Fall chinook salmon parr probably grow more slowly in present day rearing areas than they did in the relatively warmer water of the historical spawning and rearing area near Marsing, Idaho. Spring water temperatures in the Snake River near Marsing, Idaho averaged 14.2°C in 1961, 14.4°C in 1962, and 13.5°C in 1963 (Chapter Four in this report), which were the last three years this reach of river supported fall chinook salmon. Based on our regression equation, these temperatures would result in growth rates of 1.4, 1.4, and 1.2 mm/d compared to the average rates of 1.2 and 1.0 mm/d we observed in the upper and lower reaches of the Snake River from 1995 to 2000.

Although parr in present-day rearing areas might grow more slowly than their historical counterparts, they still exhibit rapid growth by comparison to those of other present-day stocks of ocean-type (Healey 1991) chinook salmon that smolt and migrate seaward as subyearlings. Growth rates reported for wild subyearling chinook salmon in presumably productive brackish and saltwater habitat along the Pacific coast of North America ranged from 0.4 to 1.3 mm/d (Healey 1980; Kjelson et al. 1982; Buckman and Ewing 1982). The rapid growth we observed occurred during the critical spring time period associated with successful smoltification (Dickhoff et al. 1997; Beckman and Dickhoff 1998), and it likely contributes to the maintenance of an ocean-type life history by most young Snake River fall chinook salmon.

We suggest that the results in this brief have an important implication for management of Snake River fall chinook salmon listed for protection under the Endangered Species Act (ESA; NMFS 1992). Management activities with the potential to decrease growth rates below 1995-2000 levels should be monitored. Releasing large numbers of hatchery fall chinook salmon into the Snake River to supplement wild production might eventually reduce growth through intra-specific competition (e.g., McMichael et al. 1997). Releasing cool water from reservoirs upstream of Lower Granite Reservoir to improve downstream migration rate and survival of fall chinook salmon smolts (e.g., Connor et al. 1998) might reduce growth by reducing water temperature. Growth rate could be used to index the effects of the above two recovery measures on the well-being of wild fall chinook salmon in the Snake River basin.

## **Acknowledgments**

Employees of the U. S. Fish and Wildlife Service Idaho Fishery Resource Office and Idaho Power Company collected data. Personnel of the Pacific States Marine Fisheries Commission maintained the PIT-tag data base. We express special thanks to T. Andersen, S. Bradbury, C. Eaton, P. Groves, A. Garcia, S. Downing, D. Marvin, R. Nelle, D. Rondorf, J. Sneva, D. Steele, C. Stein, K. Tiffan, and R. Waitt. Early drafts of the manuscript were reviewed by D. Bennett, T. Bjornn, P. Bigelow, J. Congleton, R. Graves, and K. Steinhorst. The editor, an associate editor, and two anonymous reviewers improved the manuscript. Funding was provided by the rate payers of the Bonneville Power Administration through Contract Number DE-AI79-91BP21708 administered by D. Docherty, and D. Praest.

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## CHAPTER EIGHT

Thermal Exposure of Juvenile Fall Chinook Salmon Migrating  
Through a Lower Snake River Reservoir\*

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Abstract.-Impoundment of the Snake River, Washington, has resulted in high water temperatures and late seaward migration of juvenile fall chinook salmon during summer months. To determine if juvenile fall chinook salmon are exposed to temperatures higher than the upper incipient lethal, we tagged groups of fish with temperature-sensing radio tags and tracked them in Little Goose Reservoir on the Snake River during the summers of 1998 and 1999. Spatial and temporal patterns of the reservoir's thermal environment were described using a bathythermograph. Little Goose Reservoir was generally homothermic, and temperatures selected by fish were typically not significantly different from mean water temperatures. No areas of thermal refugia existed in Little Goose Reservoir. Thermal exposure was most influenced by fish residence time in the reservoir within each year and by temperature differences between years. Current augmentation of Snake River summer flows with cold-water releases from Dworshak Dam in Idaho reduces the thermal exposure of juvenile fall chinook salmon by lowering water temperatures up to 4°C and may therefore increase their survival. Continued flow augmentation using water from Dworshak Reservoir may be the only mechanism to meet the temperature standard for the lower Snake River.

## Introduction

The construction of Ice Harbor, Lower Monumental, Little Goose, and Lower Granite dams has impounded much of the lower Snake River, Washington. The only accessible free-flowing river reach is in Hells Canyon, located upstream of Lower Granite Reservoir. The Hells Canyon Reach supports a naturally-reproducing population of Snake River fall chinook salmon (*Oncorhynchus tshawytscha*), which is listed as threatened under the Endangered Species Act (NMFS 1992). The timing of juvenile fall chinook salmon seaward migration is later than it was historically (Connor 2001), and fish currently emigrate through lower Snake River reservoirs in July and August when summer water temperatures are at a maximum.

The specific consequences of elevated thermal exposure to fish are currently unknown, but some general effects of warm water temperatures on juvenile chinook salmon are well documented. Among these is an increased risk of predation (Vigg and Burley 1991), disease (Fryer and Pilcher 1974; Holt et al. 1975), reduced gill  $\text{Na}^+/\text{K}^+$  ATPase activity (a measure of smolt physiology that has been associated with migratory behavior in chinook salmon; Zaugg 1989; Beeman et al. 1991), and low survival and detections at dams (Muir et al. 1998; Connor et al. 1998). Finally, the upper incipient lethal temperature for juvenile chinook salmon is 24°C (Brett 1952), which was routinely exceeded in the top meter of the water column in Little Goose Reservoir.

Washington has established a maximum water temperature standard of 20°C for the lower Snake River to protect salmonids pursuant to the Clean Water Act (WAC 1992). However, this standard is routinely exceeded during the summer migration of juvenile fall chinook salmon (USACE 2002). As a consequence, the U.S. Army Corps of Engineers (operators of the federal dams on the lower Snake River) is currently in litigation for allegedly violating the temperature standards in the lower Snake River (132 S. Supp. 2d. 1072 (D.OR.2000)). The only short-term method to reduce these water temperatures is to release cold water from Dworshak Reservoir. This reservoir is located on the North Fork Clearwater River 68 km upstream of the Snake and Clearwater river confluence at Lewiston, Idaho (Snake River rkm 224) (Figure 1). However, the effect of Dworshak Reservoir releases on the thermal exposure of fish in the lower Snake River is unknown because thermal history information from migrants has not been available.

Areas of cool water (thermal refugia) may exist in lower Snake River reservoirs, which would allow fish to avoid temporarily unfavorably high temperatures (Karr et al. 1998). Monitoring stations that use thermographs may not adequately describe the temperatures experienced by fish migrating through the lower Snake River because inferences are limited to specific locations. Existing reservoir temperature data are relatively sparse and were usually collected at a crude spatial scale. The most complete data sets were collected from the scroll cases at the dams (Bennett et al. 1997, Karr et al. 1998). Temperatures measured at the dams or from single thermographs in the reservoirs cannot detect the presence of thermal refugia. The thermal environment at a finer spatial scale has not been adequately described for any of the lower Snake River reservoirs.

Thermal histories of juvenile fall chinook salmon migrating through lower Snake River reservoirs are also unavailable. However, recent advances in miniature electronics have resulted in temperature-sensing radio transmitters suitable for use in fish as small as 120 mm fork length. These tags make it possible to obtain thermal histories on migrating juvenile salmon. We used a combination of radio telemetry and temperature sampling to: (1) estimate the thermal exposure of juvenile fall chinook salmon migrating through a lower Snake River reservoir, and (2) to determine if selected temperatures differed from those generally available.

### **Study Area**

We conducted this study in Little Goose Reservoir located on the lower Snake River in eastern Washington (Figure 1). The reservoir was created in 1970 with the completion of Little Goose Dam (River kilometer (rkm) 113 as measured from the Snake River mouth), and is approximately 60 km long. The reservoir has a mean width of 0.7 km and the mean depth increases from 14 to 32 m from the head of the reservoir to the forebay of Little Goose Dam. The reservoir has little storage capacity, low retention time (5.7 d at 1,416 m<sup>3</sup>/s), and is operated as run-of-the-river for hydroelectric power generation. Discharges during the summer typically decline from about 2,266 to 566 m<sup>3</sup>/s. Lower Granite Dam (rkm 173) is the only dam upriver of Little Goose Dam before the free-flowing Hells Canyon Reach, where Snake River fall chinook salmon adults spawn naturally and juveniles rear before migrating seaward.

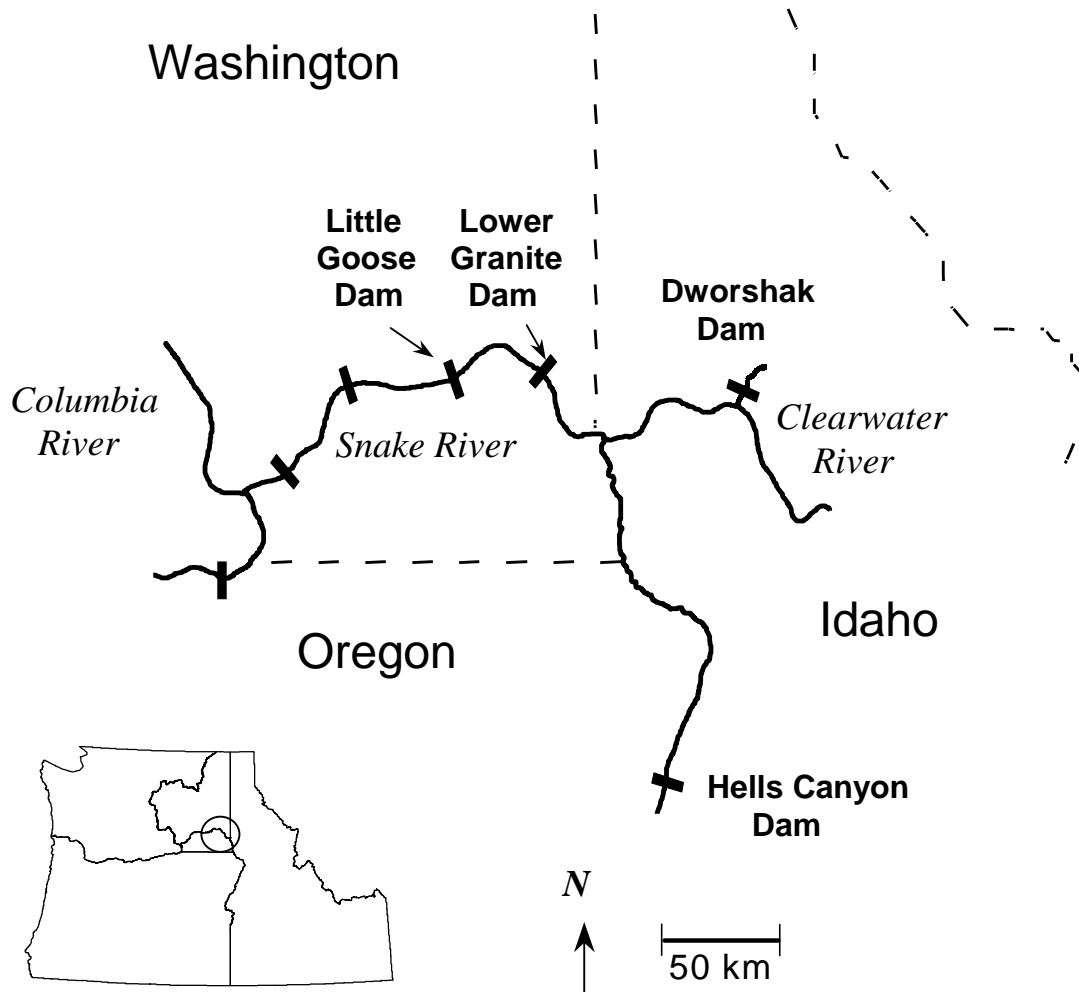


Figure 1.-Study area showing Little Goose, Lower Granite, and Dworshak dams.

## Methods

### *Reservoir temperatures*

The thermal environment of Little Goose Reservoir was characterized in 1998 and 1999 with a bathythermograph (BT) that continuously recorded time, depth, and temperature (accuracy =  $\pm 0.1^{\circ}\text{C}$ ). Twenty-five transects perpendicular to the longitudinal axis of the reservoir were established at roughly 2.5-km intervals from Little Goose to Lower Granite Dam. The BT was lowered at a rate of 0.4 m/s from the water surface to the bottom of the reservoir at five locations along each transect: south shoreline, 25% channel width (as measured from the south shore), 50% channel width, 75% channel width (as measured from the south shore), and north shoreline. Shoreline deployments were made as near as possible to the shore while staying within water 3-4 m deep. Data were collected from each transect at approximately 14-d intervals, which we refer to as time periods, from mid July to early September. For each profile, the BT temperature at the water surface was checked against the temperature measured with a mercury thermometer that met American National Standards Institute criteria. Equipment failure precluded collecting transect water temperature data during 5-19 July and 20 July-2 August.

### *Radio telemetry*

Miniature temperature-sensing radio transmitters (Advanced Telemetry Systems, Isanti, Minnesota) were used to collect thermal histories from juvenile fall chinook salmon. Transmitters were 17 mm long, 6 mm in diameter, weighed 1.7 g in air, and had a life span of 8 to 10 d. Tag accuracy was  $\pm 0.5^{\circ}\text{C}$ . The transmitters operated on unique frequencies and had pulse intervals (in milliseconds) that varied directly and linearly with temperature. The manufacturer provided equations for each tag to convert pulse interval to temperature, which were validated along with tag performance in laboratory tests (Haskell et al. 1999). Because tags did not have a temperature sensor external to the fish, measured temperatures were those of the fish at any given time. The response time of the tags implanted in fish ranged from 2 min ( $0.5^{\circ}\text{C}$  temperature change) to 4.5 min ( $6.0^{\circ}\text{C}$  change), and varied directly with the magnitude of the temperature change (Haskell et al. 1999).

To obtain thermal histories from in-river migrants, we tagged naturally-produced juvenile fall chinook salmon collected at the Lower Granite Dam juvenile fish collection facility in 1998 and 1999. Fish selected for tagging were at least 120 mm fork length (FL), had no visible signs of injury or stress, and were unmarked indicating their natural origin. Fish were anesthetized in a 100 mg/L solution of buffered tricaine methanesulfonate (MS-222), weighed, and measured. A temperature-sensing radio transmitter was gastrically implanted in the fish as described by Adams et al. (1998a). After tagging, fish were allowed to recover for 24 hr before being released into the Lower Granite Dam tailrace via the PIT-tag bypass pipe in 1998 and by boat in the middle of the channel in 1999.

Totals of 69 and 76 juvenile fall chinook salmon were tagged with temperature-sensing radio transmitters and released into Little Goose Reservoir in 1998 and 1999. The mean size of radio-tagged fish ranged from 130 mm and 24.4 g to 176 mm and 73.7 g (Table 1). The ratio of the tag weight to fish body weight averaged 3.8% (range = 2.3 to 6.9%), which was less than the 5% limit used in other studies (Winter 1983, Adams et al. 1998a, Adams et al. 1998b). Tag retention during the 24-hr recovery period was 99% (one regurgitated tag each year), and there were no mortalities.

Groups of 5-7 tagged fish were released at regular intervals over five 9 to 14-d periods each year (Table 1). The fish with the strongest radio signal from each release group was tracked continuously for 48 hr by boat as it migrated through the reservoir. Fourteen fish were chosen as primary fish and intensively tracked in 1998. The mean number of temperature records collected per fish was 3,160 (range = 203 to 9,928). In 1999, 18 fish were intensively tracked, which had a mean number of temperature records per fish of 2,772 (range = 381 to 7,960). Other fish in the release groups were monitored to provide additional temperature information. If the primary fish was not detected for 4 hr, we tracked a different fish. Radio-tag signals were detected with a data-logging receiver (Lotek Engineering Inc., Newmarket, Ontario, Canada) and a six-element Yagi antenna. Fish could only be detected to a maximum depth of 10 m due to radio signal attenuation. Each hour, the location of the selected fish was determined as the position of the strongest radio signal strength with a real-time differentially corrected global positioning system (GPS). In addition, a temperature profile of the water column was collected using a BT

Table 1.-Release dates, number of fish (N), and mean fork lengths and weights, with standard errors in parentheses, of radio-tagged juvenile fall chinook salmon released into Little Goose Reservoir in 1998 and 1999.

Release dates	N	Length (mm)	Weight (g)
<b>1998</b>			
10-19 July	14	138 (2.5)	35.4 (2.0)
20 July-1 August	13	136 (2.7)	33.3 (2.1)
2-14 August	14	142 (1.7)	38.5 (1.5)
15-28 August	14	152 (2.1)	46.7 (2.1)
29 August-7 September	14	167 (2.0)	62.7 (2.4)
<b>1999</b>			
5-19 July	17	130 (7.5)	24.4 (5.0)
20 July-2 August	11	144 (6.9)	35.5 (4.8)
3-15 August	18	154 (4.7)	44.1 (4.5)
16-25 August	12	161 (13.5)	53.5 (13.7)
26 August-4 September	18	176 (7.1)	73.7 (9.7)

to compare mean fish and water temperatures subsequently at fish locations.

Fish movements and temperature exposures were also monitored by fixed-site receivers connected to stationary antenna arrays at Rice Bar (rkm 150) in 1999 only, New York Island (rkm 125), and the Little Goose Dam and forebay as described by Venditti et al. (2000). These arrays were used to corroborate data collected during boat tracking, monitor the movements and temperature exposures of tagged fish, and monitor temperature exposures of fish in the forebay of Little Goose Dam.

### *Data Analysis*

Mean water column temperatures were calculated for each BT transect and plotted to examine longitudinal and seasonal trends. Frequency histograms were constructed to determine the amount of temperature variation within BT profiles. To determine if fish and water temperatures were different, mean hourly temperatures for each fish were compared to mean BT temperatures at fish locations using paired t tests. Only fish with five or more detections were included in the analyses. Data were analyzed by time period and year.

Thermal exposure in cumulative daily temperature units (CTU) was calculated for each fish that passed Little Goose Dam. Exposure was calculated as the time (d) from a fish's release to its first detection at an exit site at Little Goose Dam multiplied by the mean reservoir temperature for that time period. High CTUs indicate greater temperature exposure. Mean exposures were compared between time periods within each year using analysis of variance and the Student-Newman-Kuels multiple range test. Within time period comparisons were made between years using two-tailed t-tests. All analyses were conducted using SAS statistical software (SAS 2000) and significance was designated at  $P < 0.05$ .

## **Results**

### *Reservoir temperatures*

Little Goose Reservoir showed similar trends in longitudinal temperature variation in both 1998 and 1999. Water temperatures were progressively higher from the tailrace of Lower Granite Dam to the forebay of Little Goose Dam (Figure 2).



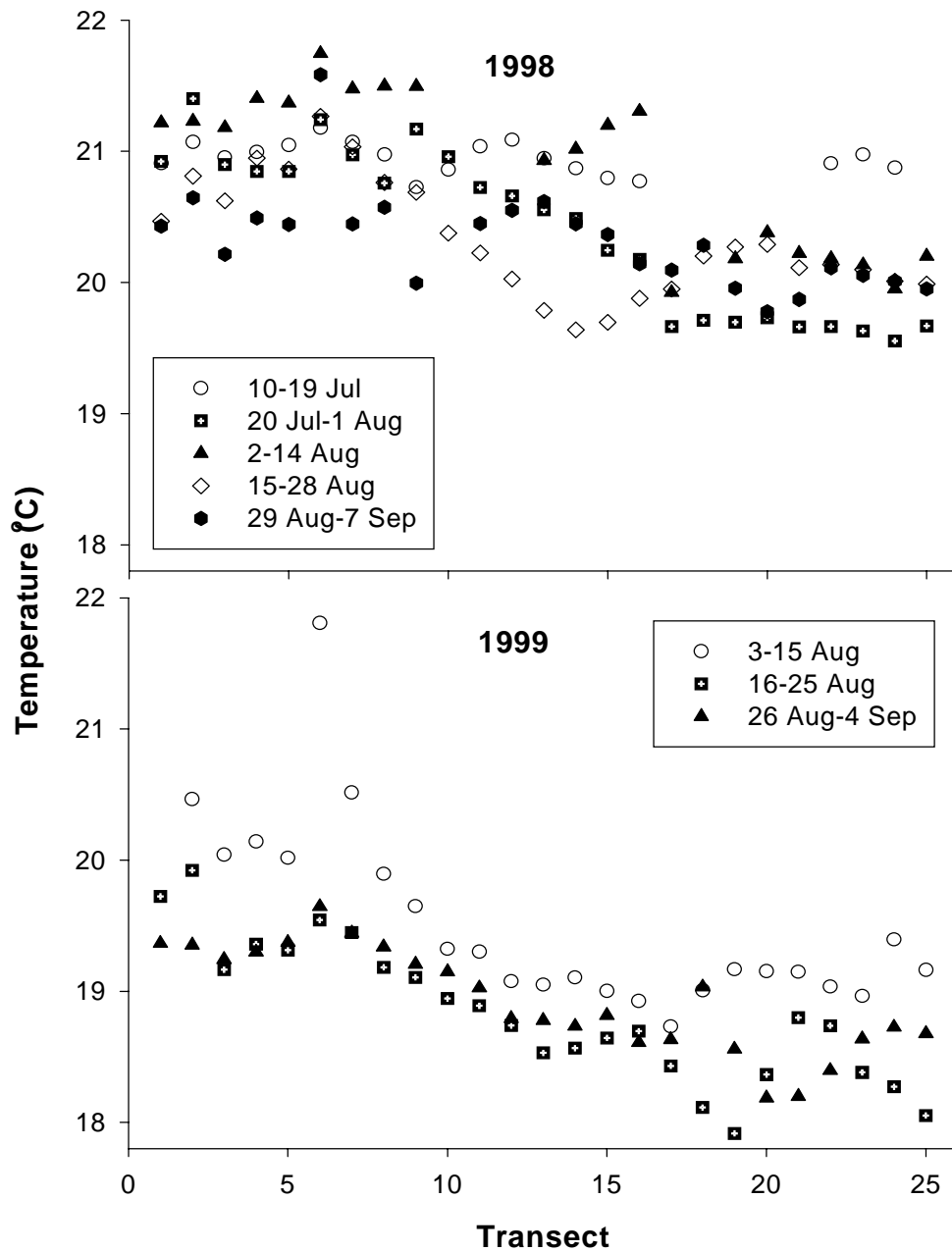


Figure 2.-Mean temperatures as measured with a bathythermograph (BT) by transect and sampling period in Little Goose Reservoir in 1998 and 1999. The solid lines represent the means of BT temperatures at each transect. Transects were spaced at approximately 2.5-km intervals and were numbered consecutively beginning with transect 1 in the Little Goose Dam forebay and ending with transect 25 in the Lower Granite Dam tailrace.

Mean BT-transect temperatures ranged from 19.5°C to 21.7°C in 1998 and from 17.6°C to 21.8°C in 1999. No individual water temperatures were <17.6°C in either year. The maximum water surface temperatures were 26.0°C in 1998 and 26.9°C in 1999. The highest water temperatures generally occurred in the top 1 m of the water column.

Mean temperatures in Little Goose Reservoir were generally >20°C in 1998 and were <20°C in 1999 (Figure 2). Seasonal variation in temperatures was greater in 1998, with the water temperatures in the upper third of the reservoir being the lowest during the latter half of July and highest in the lower half of the reservoir in early August (Figure 2). Seasonal reservoir temperatures were less variable in 1999, and the entire reservoir averaged about 0.6°C higher during the first half of August than later in the year.

BT temperature profiles from Little Goose Reservoir showed that the water column was generally homothermic with no vertical stratification, except for higher temperatures in the top 1 m. We described the vertical thermal gradients in the water column using the frequency distribution of the differences between minimum and maximum water temperatures measured in BT profiles (Figure 3). Eighty-seven percent of the BT profiles in 1998 and 82% of the profiles in 1999 had maximum vertical temperature variations of 2°C or less. The greater temperature differences in Figure 3 were due to higher water temperatures measured near the surface.

#### *Radio telemetry*

The temperature histories of radio-tagged fish in Little Goose Reservoir were generally not significantly different from mean water column temperatures measured with a BT at fish locations (Table 2). Significant differences between mean temperature of fish and water were found in 4 of 10 t-test comparisons. However, differences between means ranged from 0.0°C to 0.3°C and averaged about 0.1°C (Table 2). This mean difference is less than the error associated with the tags thus we do not consider the differences biologically significant.

We calculated the thermal exposure of 90 fish detected in the Little Goose Dam forebay or at an exit site indicating that they had passed the dam. Thermal exposure of juvenile fall chinook salmon increased throughout the summer in 1998, but no trend was evident in 1999 (Figure 4). Significant differences

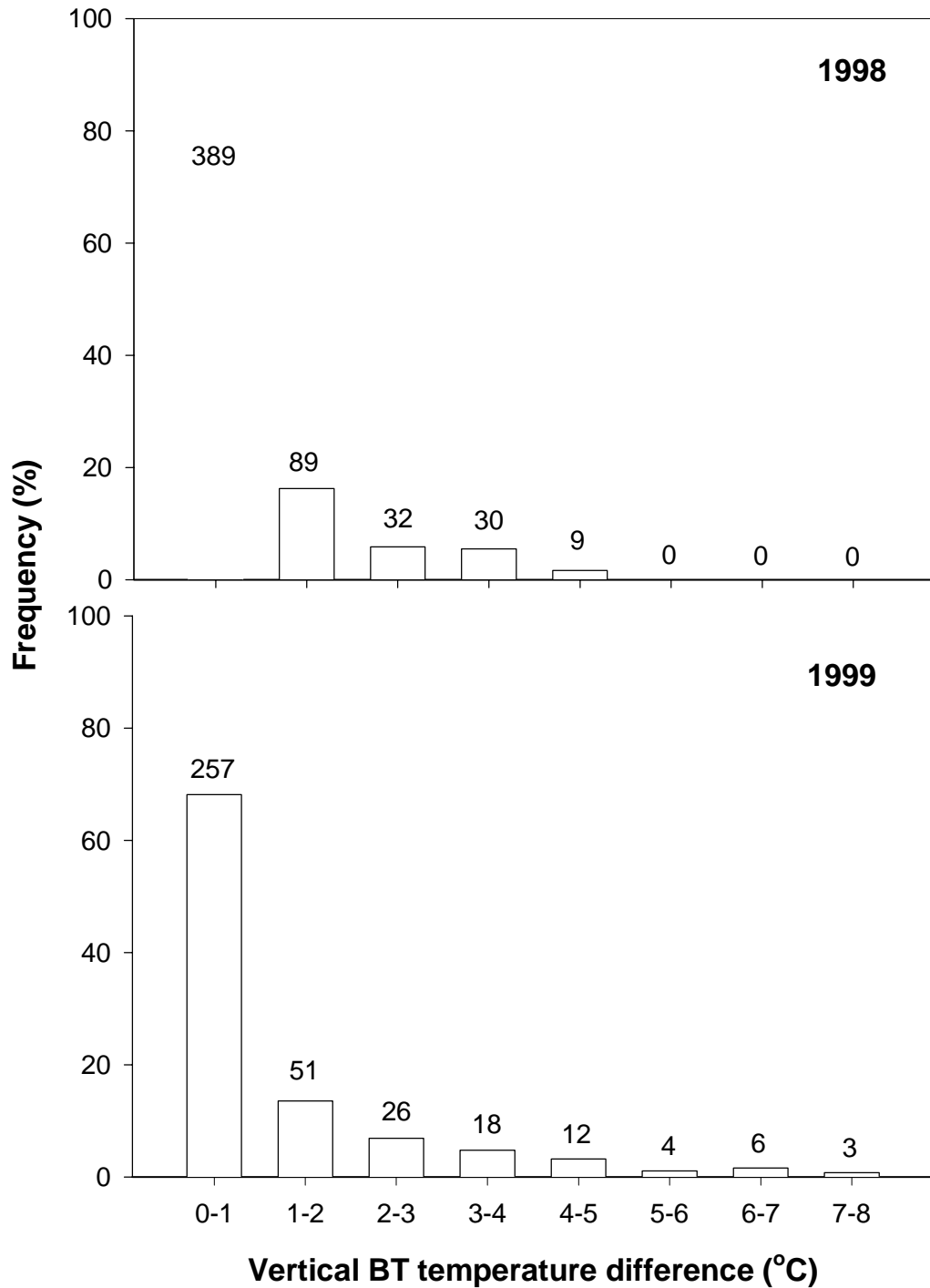


Figure 3.-Percent frequency of maximum vertical temperature differences as measured with a bathythermograph (BT) in Little Goose Reservoir in 1998 and 1999. The numbers of BT profiles in each temperature-difference category are shown above each bar.

Table 2.-Mean temperatures (°C) of juvenile fall chinook salmon fitted with temperature-sensing radio tags, and measured in Little Goose Reservoir with a bathythermograph (BT) at fish locations during 1998 and 1999. Means are summarized by time period.

Time period	N	Mean fish temp	Mean BT temp	Difference
<b>1998</b>				
10-19 July	68	20.8	20.7	0.1
20 July-1 August	12	20.9	20.8	0.1
2-14 August	41	20.4	20.3	0.1
15-28 August	8	20.6	20.5	0.1
29 August-7 September	70	20.5	20.4	0.1
<b>1999</b>				
5-19 July	46	18.2	18.0	0.2
20 July-2 August	60	19.5	19.2	0.3
3-15 August	10	19.8	19.8	0.0
16-25 August	58	19.5	19.2	0.3
26 August-4 September	93	19.4	19.3	0.1

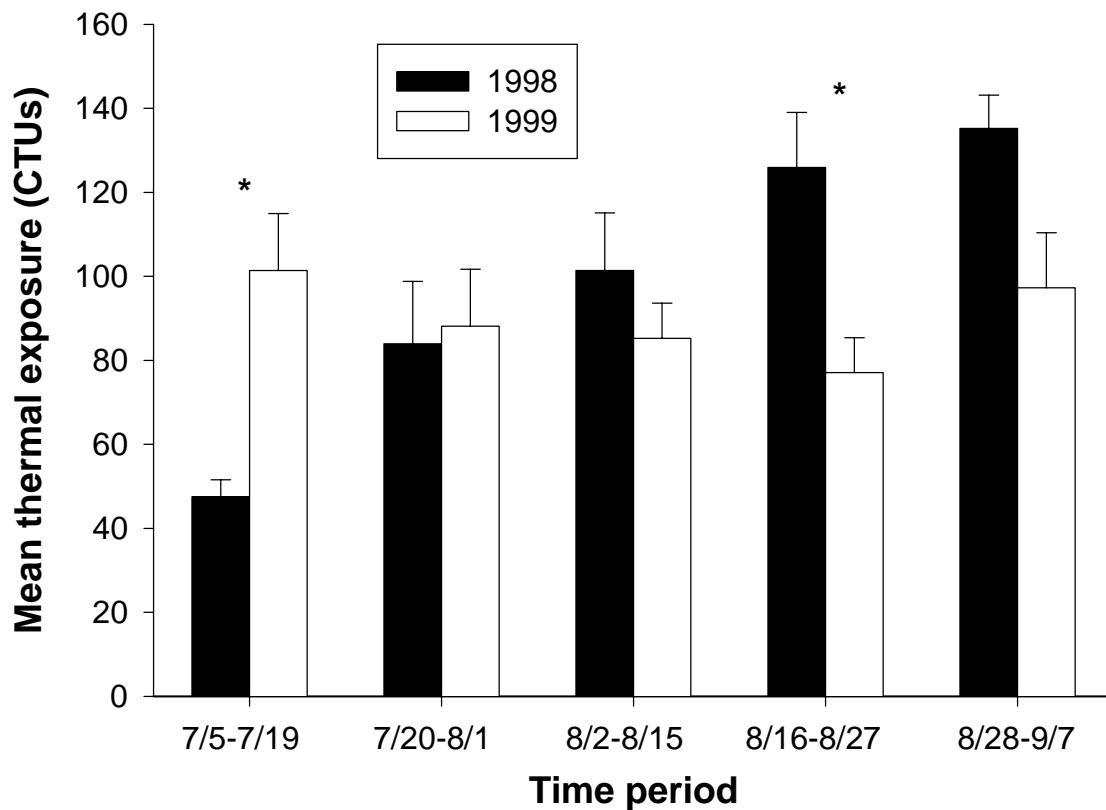


Figure 4. Mean thermal exposures of juvenile fall chinook salmon in Little Goose Reservoir by time period in 1998 and 1999. Thermal exposure is the product of fish residence time (d) in Little Goose Reservoir and the mean reservoir temperature for that time period. Standard error bars are shown. Significant ( $P < 0.05$ ) between-year comparisons are denoted with asterisks.

existed between thermal exposures for different time periods in 1998 ( $P = 0.0024$ ). The exposure of fish from the early-July time period was significantly different from fish in late August and early September periods, but there were no significant differences between exposures after late July. No significant differences in seasonal exposures were found in 1999. Between-year comparisons showed that exposures were significantly different during early July ( $P = 0.0120$ ) and late August ( $P = 0.0034$ ) (Figure 4).

## Discussion

Temperature-sensing radio tags effectively provided thermal exposure data for juvenile fall chinook salmon migrating through Little Goose Reservoir. We used gastric, rather than surgical, implantation of tags because we believed it was less intrusive for the size of fish with which we worked. Although gastric implantation reduces long-term growth in juvenile salmonids (Adams et al. 1998b), there is less potential for short-term bias in swimming performance (Moser et al. 1990, Peake et al. 1997, Adams et al. 1998a). The fish tagged in this study behaved similarly and had migration rates similar to fall chinook salmon radio tagged and released in Little Goose Reservoir by Venditti et al. (2000).

Bathythermograph temperatures in Little Goose Reservoir showed that during the summers of 1998 and 1999 the reservoir was well mixed, exhibited no thermal stratification, and had no water temperatures  $<17.6^{\circ}\text{C}$ . These data are similar to the findings of Bennett et al. (1997) who used a more coarse-scale temperature monitoring approach in the lower Snake River reservoirs in the early 1990s. Generally, fish did not have a wide range of temperatures from which to choose in Little Goose Reservoir, likely due to the low retention time of water in the reservoir, that contributes to well-mixed, homothermic conditions. Our comparisons of hourly fish temperatures and ambient water temperatures at fish locations revealed no biologically significant differences (Table 2). As a result, mean water column temperatures could be used as a surrogate for fish temperatures in Little Goose Reservoir and potentially for other main-stem reservoirs that are predominantly homothermic.

We found no evidence that thermal refugia existed in Little Goose Reservoir. We defined thermal refugia as having temperatures  $\leq 17.8^{\circ}\text{C}$ , which represents the maximum temperature preference for subyearling fall chinook salmon from the Columbia River (Sauter et al. 2001). The lowest temperature recorded by

the BT or radio-tagged fish was 17.6°C, which was measured on a single occasion in mid August, 1999.

The thermal exposure of juvenile fall chinook salmon in Little Goose Reservoir was primarily influenced by residence time within each year and by temperature differences between years. In 1998, temperatures were relatively constant over the season (Figure 2), but fish residence times in the reservoir increased seasonally. Both residence times and temperatures exhibited little seasonal variation in 1999. Although temperatures were about 1°C lower overall in 1999 compared to 1998 (Figure 2), longer residence times for fish migrating early in 1999 contributed to their greater thermal exposures. Both higher temperatures and longer residence times in late 1998 resulted in higher exposures than in 1999.

During our study, water temperatures measured at Little Goose Dam exceeded the 20°C temperature standard for 56 d in 1998 but only for 4 d in 1999. These temperatures contributed in part to the greater thermal exposure experienced by fish in 1998. Two methods to reduce thermal exposure of juvenile fall chinook salmon in the lower Snake River are 1) to reduce residence times during migration or 2) to cool the reservoir water. Since 1992, fishery managers have attempted to do both, and thereby increase survival, by augmenting summer flows with cold-water releases from Dworshak Reservoir in Idaho. Flow augmentation not only reduces water temperatures, but also increases flow, which reduces fish travel time (Berggren and Filardo 1993). Connor (2001) used a one-dimensional heat budget model to compare observed and predicted water temperatures of the Lower Granite Reservoir tailrace in 1998 and 1999 both with and without flow augmentation from Dworshak Reservoir. Figure 5 summarizes his results, which show up to a 3-4°C reduction in summer water temperatures due to flow augmentation. Without the current temperature reduction that flow augmentation affords, thermal exposure of juvenile fall chinook salmon in the lower Snake River would be higher, and would potentially further reduce their survival. Continued flow augmentation using water from Dworshak Reservoir may be the only way to prevent the exceeding the temperature standard for the lower Snake River.

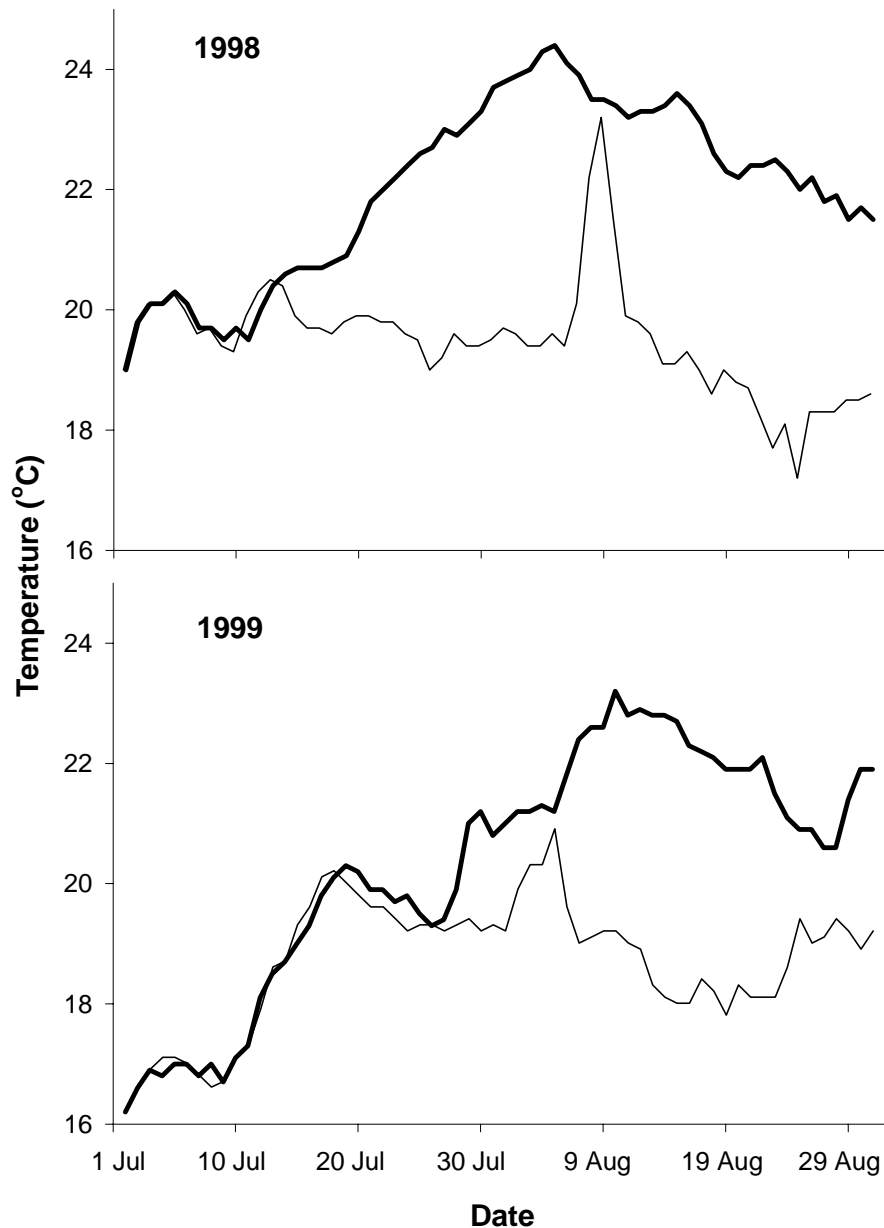


Figure 5. Mean daily observed (thin line) water temperatures in the Lower Granite Dam tailrace with flow augmentation from Dworshak Reservoir and those predicted (heavy line) without flow augmentation in 1998 and 1999. Observed temperatures were measured with a thermograph and predicted temperatures were obtained from a one-dimensional heat budget model (Yearsley et al. 2001). In 1998, flow augmentation began on 13 July and ended on 23 August, and in 1999, flow augmentation occurred from 19 July through 29 August. This figure was reproduced with permission from Connor (2001).



### **Acknowledgements**

We thank William P. Connor of the U.S. Fish and Wildlife Service for providing the data shown in Figure 5. We also acknowledge the efforts and support of our colleagues at the Columbia River Research Laboratory who participated in data collection and analysis. The critical reviews of Dena Gadowski and Tammy Salon greatly improved this manuscript. Funding was provided by the Bonneville Power Administration through contract DE-AI79-91BP21708 administered by Debbie Docherty.

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